Constructed Floodplain Wetland Effectiveness for Stormwater Management

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Abstract

A 0.2-hectare wetland was constructed in the floodplain of Opequon Creek in Northern Virginia as a best management practice (BMP) for stormwater management. The research goals were to 1) determine if wetland hydrology existed and quantify the role of groundwater exchange in the constructed wetland (CW) water budget, 2) estimate wetland hydraulic characteristics during overbank flows, and 3) quantify the event-scale nutrient assimilative capacity of the constructed wetland. CW water table elevations and hydraulic gradients were measured through an array of nested piezometers. During controlled flooding events, stream water was pumped from the creek and amended with nutrients and a conservative tracer in two seasons to determine hydraulic characteristics and nutrient reduction. Samples were collected at the inlet, outlet structure, and at three locations along three transects along the wetland flowpath.

Water table elevation monitoring demonstrated that wetland hydrology existed on the site. The mean residence time of the wetland was found to be 100 min for flow-rates of 4.25-5.1 m^3/min. Residence time distributions of the high and low marsh features identified a considerable degree of flow dispersion. Manning's *n* varied between macrotopographic features and was significantly higher in the spring event as compared to the fall event, likely due to the presence of rigid-stem vegetation. Average wetland *n* was 0.62. Total suspended solid concentrations decreased with increasing residence time during both experiments. Mass reduction of pollutants were 73% total suspended solids (TSS), 54% ammonia-nitrogen (NH_3-N), 16% nitrate-N (NO_3-N), 16% total nitrogen (TN), 23% orthophosphate-phosphorus (PO_4-P), and 37% total P (TP) in the fall, and 69% TSS, 58% NH_3-N, 7% NO_3-N, 22% TN, 8% PO_4-P, and 25% TP in the spring. Linear regression of mass flux over the event hydrograph was used to determine pollutant removal rates between the wetland inlet and outlet. Pollutant removal rates were determined through linear regression of mass flux and were higher in the spring event than in the fall. Dissolved nitrogen species were more rapidly removed than dissolved phosphorus. TSS, TP, and TN removal were greater and faster than dissolved nutrient species, suggesting that physical settling was the dominant removal mechanism for stormwater pollutants.

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Research Dissertation Outline

This dissertation is the result of research pertaining to the implementation and monitoring of a wetland for controlling stormwater pollution in the Chesapeake Bay Watershed. A synthesis of literature precedes a detailed methodology chapter and the scientific findings, which are expressed in three subsequent manuscript chapters. Conclusions and recommendations for future research in this field are reported in a final chapter. All pertinent field and laboratory logs, as well as supplemental information, can be found in the appendices. The following is an outline of the major chapters:

Chapter 1 – Introduction and Literature Review

Chapter 2 - Constructed Wetland Implementation and Methods

Chapter 3 – Ludwig, A. L. and W.C. Hession. Groundwater Hydrology of a Small Constructed Floodplain Wetland In the Ridge and Valley of Virginia. *For submission to the Journal of Hydrology*.

Chapter 4 – Ludwig, A.L. and W.C. Hession. Influence of Macrotopography on Flow Hydraulics in a Small Constructed Floodplain Wetland. *For submission to the Journal of Water Research*.

Chapter 5 – Ludwig, A.L. and W.C. Hession. Event-Scale Nutrient Attenuation in a Small Constructed Floodplain Wetland. *For submission to the Journal of Environmental Quality*.

Chapter 6 - Conclusions and Future Research Recommendations

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List of Abbreviations

BMP – Best management practice

Br – Bromide

cms – cubic meter(s) per second

C – Carbon

CSTR – Continuously stirred tank reactors

CW – Constructed wetland

DOC – Dissolved organic carbon

DRP – Dissolved reactive phosphorus

FWS – Free water surface

g – Gram(s)

ha – Hectare(s)

HGM – Hydrogeomorphic

hr – Hour(s)

HRT – Hydraulic residence time

HSSF – Horizontal subsurface flow

ICW – Integrated constructed wetland

kg – Kilogram(s)

km – Kilometer(s)

KBr – Potassium bromide

L – Liter(s)

LOI – Loss on ignition

m – Meters(s)

mg – Milligram(s)

min – Minute(s)

mL – Milliliter(s)

mm – Millimeter(s)

N – Nitrogen

NPS – Non-point source

NH_3-N – Ammonia as nitrogen

NO_3-N – Nitrate as nitrogen

NRCS – Natural Resources Conservation Service

P – Phosphorus

PFR – Plug-flow reactors

PO_4-P – Orthophosphate as phosphorus

OP – orthophosphate

s – Second(s)

SRP – Soluble reactive phosphorus

TIS – Tanks in series

TP – Total Phosphorus

TN – Total Nitrogen

TOC – Total organic carbon

TSS – Total suspended solids

USACE – United States Army Corps of Engineers

USEPA – United States Environmental Protection Agency

USGS – United States Geologic Survey

VA - Virginia

VADCR – Virginia Department of Conservation and Recreation

VF – Vertical flow

VSS – Volatile suspended solids

IFI – Inter-flood interval

I. Introduction and Literature Review

On May 12, 2009, President Barack Obama declared the Chesapeake Bay a national treasure and ordered the beginning of *"…a new era of shared Federal leadership with respect to the protection and restoration of the Chesapeake Bay*." In his order, President Obama identified nutrient pollution as a leading cause of impairment to the Chesapeake Bay, which does not meet the "fishable and swimmable" standards mandated by the Clean Water Act (1972).

Through this order of immediate action, President Obama has charged researchers to provide guidance for innovative and adaptive management practices for water quality improvement. As highlighted, stormwater is a targeted source of pollutants. Scientists and engineers must act upon this order by providing research on the effectiveness and capacity of management practices in fulfillment of the goals outlined in the Presidential order. The Chesapeake Bay Protection and Restoration order included "recommendations" for federal agencies in accomplishing a series of "*steps to protect and restore the Chesapeake Bay."* The following are steps relevant to stormwater management:

"(a) define the next generation of tools and actions to restore water quality in the Chesapeake Bay and describe the changes to be made to regulations, programs, and policies to implement these actions;

…(c) strengthen storm water management practices at Federal facilities and on Federal lands within the Chesapeake Bay watershed and develop storm water best practices guidance;

…(f) strengthen scientific support for decision making to restore the Chesapeake Bay and its watershed, including expanded environmental research and monitoring and observing systems; …"

The Chesapeake Bay is just one example of the need for protection of our national water resources. The annual cost of eutrophication in the United States (US) has been estimated as being approximately $2.2 billion dollars (Dodds et al., 2008); much of this cost s a result of lost recreational area and associated decrease in property values. In a report to Congress, the US Environmental Protection Agency (USEPA) reported that only 56% of the nation's waters were meeting the water quality standards for their designated uses (USEPA, 2009). This poses the need for engineered solutions to eutrophication through nutrient attenuation from surface waters.

Due to the continual challenge of increasing population and changing landscapes, adaptive management practices for water-quality improvement continue to be a much-needed area of research. In a review of evidence, Zedler (Zedler, 2003) aligned the loss of wetlands in the Midwestern US to the loss of ecological services that wetlands provide, including flood abatement, water quality improvement, and biodiversity support. Zedler (2003) also evaluated value-placed ecosystem service analysis findings reported by Costanza (Costanza et al., 1997) related specifically to those services performed by wetlands and found that 39.6% of the total global ecological services could be attributed to the functions of wetlands. With the provision of such beneficial services, wetland creation and restoration is increasingly being used to help mitigate some of the nation's water quality problems (Mitsch and Gosselink, 2000).

Created wetlands are often used for stormwater treatment; however, new solutions to pollutant removal call for the use of these practices in innovative new ways and in new landscape settings throughout the stream network. The overall goal of this work was to evaluate the capacity of a created floodplain wetland to attenuate sediments and nutrients for stormwater management. While much of the results of the work are site-specific ecological phenomena, the research activities involved here – the design, construction, and monitoring – were performed to support the concept that floodplain wetland creation may be a viable management practice for the reduction of stormwater pollutants in surface waters. The central objectives of this work were to:

1. Design and construct a floodplain stormwater treatment wetland using an integrated approach that calls for minimal artificial inputs during and after construction to facilitate pollutant removal.
2. Characterize the wetland hydrology in the created floodplain wetland

3. Determine the role of macrotopography and vegetation on flow mixing during controlled flow-through events.
4. Evaluate the capacity of the floodplain wetland to remove stormwater pollutants at the event time scale during controlled flow-through conditions.

The broader impacts of these findings lie in the demonstration of floodplain wetland creation, identification of wetland hydrology, and evaluation of event-scale nutrient removal within this built environment. Adaptations unique to this management practice include the landscape placement and application of the practice (in the floodplain), the theme of self-sustainability of the system that was incorporated into the design, and the design qualities that benefitted the landowner, as well as facilitated water-quality improvement and habitat creation.

Literature Review

I.1 Stormwater

Sources of Stormwater

In-stream nutrient concentrations have been correlated with human activity in their corresponding basins (Gergel et al., 2002). Agricultural sources of nutrient pollution are correlated with farming practices. Excessive application of fertilizers or applications proximate to heavy rains may cause nutrients to be transported in runoff from uplands and floodplains to streams (Stone et al., 2003). Further contributions from the streambed of dissolved nitrogen (N) and phosphorus (P) through re-mineralization also increase concentrations of water column N and P (Ensign et al., 2006). The urban setting also provides a wide variety of sources of stormwater runoff due to the impervious nature of much of the infrastructure that inhabits urban areas – rooftops, parking lots, streets, highways, landscaping, lawns, and industrial parks. In a summary of reported pollutant concentrations contributed from urban sources, the leading contributor of non-point source (NPS) nutrient pollution was urban lawns (Kadlec, 2009a).

Effects of Eutrophication

The effects of eutrophication were estimated to cost $2.2 billion dollars annually in the US, mainly due to the loss of property value and recreational areas (Dodds et al., 2008). These effects were linked to TP and TN concentrations, which were found to be over 90% higher in the rivers of 12 out of 14 ecoregions than in reference river segments. In their 2004 Report to Congress, the United States Environmental Protection Agency (USEPA) reported only 56% of the nation's surface waters were of a quality that sustained their designated use(USEPA, 2009). The leading causes of impairment were identified as pathogens, habitat alterations, and organic enrichment/oxygen depletion. Leading sources of impairment were identified as agricultural activities and hydrologic modifications (i.e. diversions or channelization). Over 30% of the assessed river miles were identified as impaired by nutrients and/or sediment. Riparian wetlands have been identified as important landscape features for the management of nutrients reaching receiving waterbodies, such as drinking water supply reservoirs and ecologically sensitive estuaries (Mitsch et al., 2001).

I.2 Stormwater Management Practices

On-site Management

On-site management for nutrients and sediment control refers to the implementation of strategies for infiltrating and treating runoff before it reaches channelized flow. These practices include but are not limited to infiltration basins, bioretention areas, rain gardens, porous pavement, rainwater collectors, and grass buffer strips. The effects of urbanization on soil structure often cause infiltration and bio-treatment potentials to diminish due to compaction (Pitt et al., 2002). Studies on effective media for infiltration and bioretention identify favorable characteristics for the treatment of pollutant-laden waters (Davis et al., 2001; Hsieh and Davis, 2005; Hunt et al., 2006).

Off-site Management

Traditional detention ponds perform the function of slowing down stormflow; however, these practices did not have any nutrient-assimilative design components. Detention pond retrofitting has become increasingly more prevalent in urban landscapes, where dense development or polluted receiving waters have increased the need for municipalities and governments to maximize the pollutant-removal capacity of areas already designated as best management practices.

Constructed wetlands of many variations have become popular BMPs for NPS pollution treatment due to their cost-effectiveness. BMPs that incorporate the design principles and removal mechanisms of constructed wetlands include wet ponds, pocket wetlands, and constructed floodplain wetlands. These BMPs vary in source water and hydrogeomorphic (HGM) positioning within the watershed, and are described based on these controls. Pocket wetlands describe those areas that may be small in size and located in tight urban development. These “pockets” of land are identified as areas to which stormwater flows and may be detained or retained. During detention/retention, pollutant-laden stormwater is slowed down and contact time is increased to maximize the nutrient assimilative capacity of the area.

Floodplain Management

Floodplains offer a suite of characteristics that facilitate hydraulic and nutrient retention, such as wetland vegetation, low slope gradients, and potential for connectivity to the stream network (Bradley, 2002). These characteristics make floodplains desirable locations for wetland creation by enhancing connectivity to the stream and, subsequently, the sorption and immobilization processes performed by wetland vegetation, microbes, and soils found in riparian areas (Tockner et al., 2010). Creating access for overland flow or overbank storm flows from adjacent stream channels within constructed wetlands in floodplains offers yet another setting for treatment to occur.

The capacity of a CW to remove pollutants from stormwater is a function of site-specific physical and chemical characteristics of wetland substrates (Carleton et al., 2000; Kincanon and McAnally, 2004; Reddy et al., 1999) and the pollutant delivery mechanisms and hydrology of the area (Braskerud, 2002a; R.H. Kadlec, 2009). The characteristics that affect wetland nutrient removal capacity must be considered during design, construction, and management of these BMPs (Fisher and Acreman, 2004; Kadlec and Hey, 1994). The concept of addressing site-specific characteristics in constructed wetland implementation has been termed the 'Integrated Constructed Wetland' (ICW) concept (Harrington and McInnes, 2009). The ICW approach explicitly incorporates three objectives: 1) water quantity and quality management, 2) utilizing landscape-fit characteristics towards improving aesthetic values, and 3) enhancement of biodiversity.

Measuring Treatment Performance

Monitoring and maintenance of treatment wetlands used for pollutant removal is an area of research that is becoming increasingly more important in the field of water quality management. As traditional treatment facilities age and new innovations in the area of low impact development are being incorporated into our landscapes, it is vital that the way in which we monitor the efficiency of these practices at removing pollutants is essential for continued improvements in their use. A well thought-out maintenance plan is also critical for perpetuation of optimal BMP performance.

There are many factors that affect treatment performance and how it is determined. Such factors include, but are not limited to; number of storm samples, computation methods for determining pollutant removal efficiency, monitoring techniques, internal geometry and storage volume, sediment/water column interactions, regional soil and climactic variations, latitude, and contributing watershed characteristics (CWP, 2007).

There exist wide variations in treatment performance published in literature. Generally, higher reductions are published from studies that dealt with large nutrient loadings and more variability in reduction is associated with lower nutrient loadings. This variability and trend is expressed by the collection of reported percent phosphorus removal and loadings listed in Table 1.1.

Table I.1
Summary of constructed wetland studies for pollutant removal purposes in a variety of applications.

Type of Wetland	Location	Field Methodology	Percent Removal (%) (constituent)	P Inflow Concentration (constituent)	Citation
Restored Forested Riparian	Georgia, USA	Groundwater well, runoff, outlet	66 (DRP) 66 (TP)	1.37 mg/L (DRP) 1.48 mg/L (TP)	(Vellidis et al., 2003)
Constructed Agricultural	Ireland	Inlet/outlet, intermediate stations	5-84 (SRP)	18.8 mg/L (SRP) or 130 g/d	(Dunne et al., 2005)
Constructed Agricultural (ICW)	Ireland	Inlet/outlet	99.7 98.2 81.4 92.9 98.3 98.8 97.2 96.2 99.6 92.0 99.0 93.3	75.7 mg/L (TRP) 15.5 18.1 22.8 14.3 10.8 1.5 11.6 5.3 12.0 43.7 0.9	(Harrington and McInnes, 2009)
Constructed Agricultural FWS	Ireland	Inlet/outlet, piezometer	92	11.5 mg/L (DRP)	(Moustafa et al., 1998) (Mustafa et al., 2009)
Constructed Agricultural Subsurface	Taiwan	Inlet/outlet	47-59 (TP)	39-44 mg/L (TP)	(Lee et al., 2004)

Type of Wetland	Location	Field Methodology	Percent Removal (%) (constituent)	P Inflow Concentration (constituent)	Citation
Constructed Agricultural	Kansas, USA	Grab	73 (TP) 38 22 18 76 (PO_4) 6 20 -132	11 mg/L (TP) 33 4 11 5.4 (PO_4) 4.4 1.3 1.7	(Mankin and Ikenberry, 2004)
Constructed Agricultural	United Kingdom	Mesocosm	84 (TP)	1.0 mg/L (TP)	(Mankin and Ikenberry, 2004)
Constructed Agricultural	Maryland, USA	Inlet/outlet	96 (TP) 84 (PO_4)	52.6 mg/L (TP) 56.6 (OP)	(Schaafsma et al., 1999)
Constructed Agricultural Subsurface	Nova Scotia	Inlet/outlet	62-96 (TP)	44.2 mg/L (TP) or 1.3 kg/ha-d (TP)	(Smith et al., 2006)
Constructed Agricultural	Nova Scotia	NA	54 (TP) 53 (SRP)	48 mg/L or 1.5 kg/ha-d (TP) 39 mg/L or 1.0 kg/ha-d (SRP)	(Wood et al., 2008)
Constructed Agricultural FWS	Australia	Inlet/outlet surface water	-20 to 45	NA	(Raisin et al., 1997)
Riparian Forested	Maryland, USA	Stream discharge, groundwater, overland flow	84 (TP) -4 (TDP)	0.435 kg/ha (TP) or 4.82 mg/L (TP) 0.208 mg/L (TDP)	(Peterjohn and Correll, 1984)
Riparian	North Carolina, USA	Soil Incubations, Surface Water	50	0.012-0.051 mg/L (PO_4)	(Cooper et al., 1987)
Riparian	North Carolina, USA	Runoff	60 (TP) 50 (SP)	NA	(Daniels, 1996)
Floodplain FWS	Maryland, USA	Inlet/outlet, intermediate stations	-41 9 10 -8	0.048 mg/L (TP) 0.073 0.10 0.096	(Noe and Hupp, 2007)
Natural Riparian	South Carolina, USA	Groundwater, inlet/outlet surface water	010-100 95-100 98-100 80-100 41-100 0-100 8-100 99-100 90-100 0-100 17-100	1 g (PO_4) 21 93 91 311 7 6 4156 105 2-5 54	(Casey and Klaine, 2001)
Constructed Floodplain	Florida, USA	Inlet/outlet, intermediate stations	72	0.052 mg/L (TP) or 0.5 g/m^2-y	(Moustafa et al., 1998)

Type of Wetland	Location	Field Methodology	Percent Removal (%) (constituent)	P Inflow Concentration (constituent)	Citation
FWS	Florida	NA	44 (TP) 45 (PP)	0.27 mg/L (TP) 0.011 (PP)	Unpublished Data ***
FWS	Norway	Inlet/outlet	37 (TP) 45 (PP)	0.3 mg/L (TP) 0.234 (PP)	(Braskerud, 2002b)
FWS	Texas	NA	49 (TP) 41 (PP)	0.87 mg/L (TP) 0.234 (PP)	Unpublished Data ***
FWS	Australia	NA	54 (TP) 68 (PP)	0.92 mg/L (TP) 0.07 (PP)	Unpublished Data ***
FWS	California	NA	37 (TP) 73 (PP)	1.30 mg/L (TP) 0.44 (PP)	Unpublished Data ***
FWS	California	NA	50 (TP) 64 (PP)	1.40 mg/L (TP) 0.17 (PP)	Unpublished Data ***
FWS	Kentucky	NA	8 (TP) 41 (PP)	4.89 mg/L (TP) 1.78 (PP)	NADB database (1998)***
FWS	Kentucky	NA	16 (TP) 68 (PP)	4.81 mg/L (TP) 1.41 (PP)	NADB database (1998)***
FWS	Australia	NA	3 (TP) 5 9 (PP) 54	9.65 mg/L (TP) 9.65 1.55 (PP) 1.55	Bavor et al., 1998***
Floodplain	Taiwan	Inlet/outlet, intermediate stations	53.3 52.8 54.0 71.0 85.1 66.1 38.3 19.6	2.27 mg/L (OP) 3.38 0.87 1.14 1.01 2.10 3.97 3.27	(Jing et al., 2001)
Riparian	Illinois	Inlet/outlet	62-92 53-90	0.011-0.04 mg/L (TP) 0.012-0.057	(Mitsch, 1995)
Urban Stormwater	Washington, USA	Inlet/outlet, groundwater, precipitation	7.5 82.4	0.075 mg/L (TP) or 117.5 kg/yr 0.030 mg/L (TP) or 53.9 kg/yr	(Reinelt and Horner, 1995)
Urban Stormwater	Virginia, USA	NA	14.9 23.6	0.35 mg/L (TP) 0.17 mg/L (OP)	(Carleton et al., 2001)
Urbaon Stormwater	Virginia, USA	Runoff, Inlet/outlet	45.9 35.8	0.14 mg/L (TP) 0.06 mg/L (OP)	(Carleton, 2000)
Floodplain Pumped	Ohio, USA	Inlet/outlet	59 70	0.017 mg/L (DRP) 0.169 (TP)	(Nairn and Mitsch, 1999)

*** Data from Kadlec and Wallace (2009). TP – total phosphorus, TRP – total reactive phosphorus, PO_4 – ortho-phosphorus, PP – particulate phosphorus, SP – soluble phosphorus, DRP – dissolved reactive phosphorus, OP – orthophosphate, SRP – soluble reactive phosphorus.

The most widely used method for determining BMP treatment performance is the percent removal (PR) approach. This approach compares the water quality at the outlet to that of what is coming into the practice either on the basis of pollutant concentration or total mass flux. Reporting the total mass flux of constituents requires the quantification of flow at both the inlet and outlet and over time. While a challenge in the field, this is the most descriptive way to quantify treatment performance. Mass reduction is the difference between the inlet and outlet flux. Equations used in published studies on treatment performance are presented in Table 1.2.

Table I.2
Summary of evaluation methods for pollutant removal performance of treatment wetlands.

Citation	Performance Evaluation Methodology
(Casey and Klaine, 2001)	Percent attenuation as described by nutrient and tracer injections; $$\%Attenuation = \left(1 - \frac{\frac{C_o}{Br_o}}{\frac{C_i}{Br_i}}\right) \times 100\%$$ C_o – observed constituent concentration, Br_o – observed tracer concentration, C_i – initial constituent concentration, Br_i – initial tracer concentration
(Smith et al., 2006)	Removal; $$R = \frac{C_i - C_o}{C_i} \times 100\%$$ Mass reduction; $$MR = \frac{\left(\overline{C_i} x \sum Q_i\right) - \left(\overline{C_o} \times \sum Q_o\right)}{\left(\overline{C_i} \times \sum Q_i\right)} \times 100\%$$ C_i – inlet concentration, C_o – outlet concentration, Q_i – inflow, Q_o – outflow, R - removal

Citation	Performance Evaluation Methodology
(Moustafa et al., 1998)	Daily load; $Load_{daily} = Q \times C$ Observed Change; $Change = M_1 - M_2$ Residence time; $RT = \frac{M_{tot}}{t_r}$ Q – flowrate, C – constituent concentration, M_1 – mass at t=1, M_2 – mass at t=2, RT – residence time, M_{tot} – total mass, t_r – replacement rate
(Kadlec, 2009a)	Event mean concentrations; $EMC = \frac{\sum(VC)}{\sum V}$ Concentration reduction; $CR = \frac{(EMC_i - EMC_o)}{EMC_i}$ Load Reduction; $LR = \frac{(V_i EMC_i - V_o EMC_o)}{V_i EMC_i}$ V – water volume, C – concentration, EMC
(Kadlec, 1996)	Phosphorus Removal, first order rate model; $C_{out} = C_{in} e^{\left(\frac{-k}{q}\right)}$ C_{out} – concentration at outlet, C_{in} – concentration at inlet, k – removal rate constant, q – hydraulic load

Citation	Performance Evaluation Methodology
(Braskerud, 2002b)	Phosphorus Removal; $$C_{TP_{out}} = 0.048 + 0.55C_{TP_{in}} - 0.014q$$ C_{TPout} – outlet TP concentration, C_{TPin} – inlet TP concentration, q – hydraulic loading rate
(Dortch, 1996)	Removal efficiency; $$1 - \frac{RE}{100} = e^{k_v \tau} \qquad 1 - \frac{RE}{100} = e^{-k_a / q}$$ RE – percent of inlet concentration or mass retained, τ - retention time, q – hydraulic loading, k_a and k_v – first order rate constants
(Reinelt and Horner, 1995)	Estimated Event Mean Concentrations; $$EMC_{TSS} = a_o + a_1 I + a_2 D + a_3 P + a_4 Q$$ $$EMC_{est} = b_o + b_1 EMC_{meas}$$ EMC – event mean concentration, a and b – regression coefficients, D – precipitation duration, I – precipitation intensity, P – preceding week precipitation intensity, Q – mean daily flow rate

Pollutant loading trends also significantly affect the performance and interpretation of efficiency computations (Schueler and Holland, 2000a). If input concentrations are near the "irreducible level" for a constituent (Schueler and Holland, 2000b), then estimates of pollutant removal may be artificially low just based on the fact that the input loading is low. This would provide no indication of what would occur at higher pollutant levels.

Created wetland designs depend on assumptions made to address the inherent complexities of the natural systems they are trying to mimic. To achieve optimal performance and inform better design, the magnitude and scale at which these assumptions are used must be minimized through research findings. Tools such as surface area-to-volume ratios and depth ratios are used as rules of thumb for the design process to introduce complexity and heterogeneity while allowing these rules to be flexible for site-specific circumstances.

I.3 Treatment Wetlands

Wetland Definition

Wetlands are found in many different landscapes and possess unique hydrology, soils, and vegetation. In the US, the Army Corps of Engineers (USACE) has jurisdiction over wetlands that have an established *nexus* to waters of the US. A significant nexus exists when it is demonstrated that the wetland has "more than a speculative or insubstantial effect on the chemical, physical, and biological integrity of a tranditional navigable water (USACE, 2007)." Jurisdictional wetlands are delineated as areas that contain hydric soils, established wetland vegetation, and wetland hydrology, which is determined through the observation of the water table ponded or flooded at the surface of the wetland for a duration of the growing season. The USACE defines wetlands as *"those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions."* (USACE, 1987)

Riparian Wetlands

Wetlands can be classified based on their function and position within the landscape. The HGM method classifies wetlands into groups based on three factors: 1) geomorphic setting within the landscape(Brinson, 1993); 2) source water; and 3) hydrodynamics. Riparian (or floodplain) wetlands are a class of wetlands established where surface topography meets the water table (Mitsch and Gosslink, 2002). Riparian wetlands are of particular interest with regard to water quality because of their connectivity to flowing waterways. Designed riparian wetlands are fed primarily by a flooding stream channel and allow stormflow to deposit sediments and chemicals on a seasonal basis into the floodplain (Mitsch, 1992b). Topography and channel morphology dictate the manner in which water enters and leaves a riparian wetland, as well as inundation frequency and duration (Fig. 1.1).

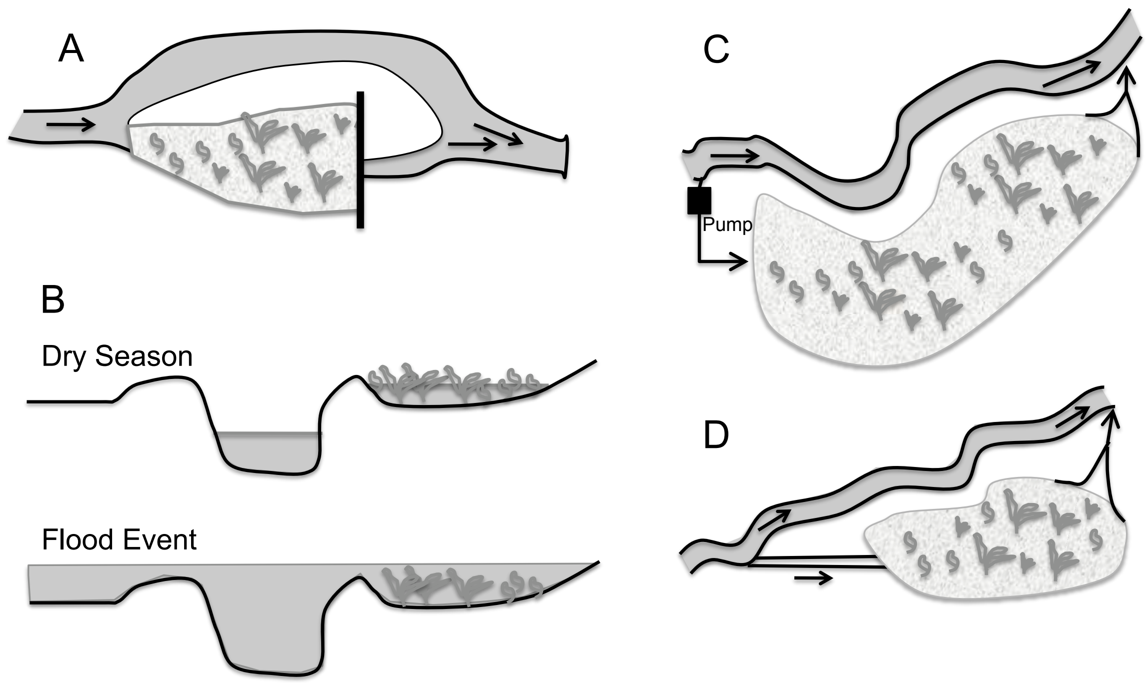

Figure I.1 Landscape placement for natural and constructed wetlands for non-point source pollution management (Mitsch, 1992b). A – Constructed in-stream wetland with high-flow bypass; B – Natural or constructed wetland fed by flood events; C – Riparian wetland fed by pumping; and D – riparian wetland fed by gravity from upstream flow diversion.

Fluctuations in hydrologic drivers in these riparian systems create a dynamic environment, providing wide heterogeneity in ecosystem services and biodiversity (Burt and Pinay, 2005; McClain et al., 2003). These environments act as hotspots along the stream network for biogeochemical processing due to the presence of meso- and micro-scale energy gradients in redox potentials and continual nutrient loading from contributing flows (Burt and Pinay, 2005; Tockner et al., 1999). Riparian wetlands and corridors have many benefits such as: 1) reducing erosion (Barling and Moore, 1994), 2) storing and processing of organic matter (Mulholland, 1997), 3) removing nutrient pollution from agricultural runoff (Vellidis et al., 2003), 4) retaining flood pulses (Tockner and Stanford, 2002), 5) creating wildlife habitat (Seavy, 2008), and 6) facilitating the establishment of wetland vegetation that is beneficial for nutrient removal (Pollock et al., 1998).

Types of Treatment Wetlands

Treatment wetlands are built environments that are designed to exploit specific characteristics of wetland ecosystems for the purpose of improving water quality. Three main categories of treatment wetlands are free water surface (FWS) wetlands, horizontal subsurface flow (HSSF) wetlands, and vertical flow (VF) wetlands (Kadlec, 2009a). Each category employs a combination of layout, media, plants, and flow patterns that are suitable for handling the designed hydraulic load and targeted pollutants. FWS wetlands are typically used for treatment of non-point source pollution in irregular or pulsed flows due to their ability to store and retain variable volumes of storage (Kadlec, 2009a).

Riparian wetlands may be used as stormwater treatment wetlands and have been identified as important landscape features for the management of nutrients reaching receiving waterbodies, such as drinking water supply reservoirs and ecologically sensitive estuaries (Mitsch et al., 2001). Efforts have been made in the field of floodplain and wetland restoration to hydrologically reconnect riparian areas with impacted stream channels with the goal of recovering the ecological services provided by riparian wetlands (Acreman et al., 2003). Non-point source pollutants have shown to be attenuated from flows through natural and constructed floodplain wetlands in the time scale of natural storm events (Casey and Klaine, 2001; Noe and Hupp, 2007; Schulz and Peall, 2001).

Factors Affecting Performance of Treatment Wetlands

The driving pollutant removal mechanisms in a treatment wetland are physical settling of solids, chemical sorption of dissolved constituents, and biological uptake and immobilization by microbes and wetland vegetation (Kadlec, 2009a). The time scale at which these mechanisms occur varies dependent on seasonal factors such as temperature and solar energy inputs as well as other variables such as hydrologic and chemical inputs. Studies have also determined that other factors impact treatment performance of wetlands, such as wetland area to watershed area ratios (Carleton et al., 2001), vegetation establishment and processes (Neubauer et al., 2005b; Reddy et al., 1999), and antecedent conditions between input events (Kadlec, 2010).

Seasonal Variability in Treatment Performance

Seasonal differences in hydraulic loading have also been shown to affect treatment performance in wetlands by decreasing nutrient retention (Raisin et al., 1997). Pollutant removal mechanisms in a wetland are subject to changing environmental factors on daily and seasonal time scales. Nitrogen dynamics vary within a wetland, where N accumulates in the subsurface during seasonally low water table periods and minimal biotic uptake (Cirmo and McDonnell, 1997). P uptake by macrophytes and periphyton is at its peak during the growing season, and seasonal release of P in the fall months may occur during decomposition and water table drawdown (Pant and Reddy, 2001).

Treatment Wetland Design

Constructed wetlands are designed to create an environment that fosters the processes found in naturally occurring wetlands and have been shown to provide similar nutrient-assimilative functions (Carleton et al., 2001; Mitsch, 1992a; Moustafa et al., 1996; Noe and Hupp, 2007). Optimization of these functions through design makes constructed wetlands useful tools in pollution mitigation (Harrell and Ranjithan, 2003). Based on extensive amounts of research in wetlands, Mitsch (1992b) reported preliminary wetland design principles:

- Use concepts of self-design to minimize required maintenance.
- Design to utilize advantageous natural energy gradients, such as storm pulses
- Design with the landscape and expectations of natural temporal variability
- Identify major and secondary design objectives
- Design the system as an ecotone, a transitional area between the uplands and stream
- Expect the system to take time to establish
- Design the system to function in order to accomplish the objective, not to obtain a preconceived form
- Mimic natural systems and do not over-engineer

Wetland Hydrology

Wetland hydrology is created through the interaction of components of the hydrologic cycle within the wetland area (Fig. 1.2). These components may be put into perspective of each other in the form of a wetland water budget. Input components of a wetland water budget include precipitation, surface and subsurface inflow, and discharge groundwater. Output components include evapotranspiration (ET), surface and subsurface outflow, and recharge groundwater. The variation in magnitudes over annual cycles affect wetland storage dynamics and influence wetland hydroperiods, or the short-term fluctuations in the water table (Richardson and Vepraskas, 2001).

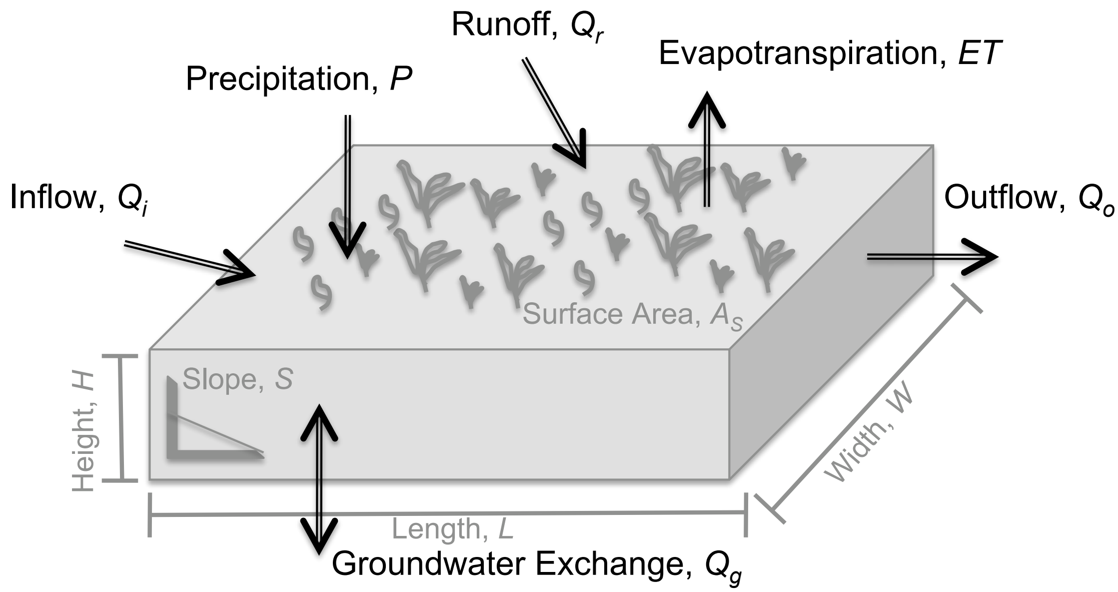

Figure I.2 Wetland water budget components (Kadlec, 2009a).

The hydrology of a riparian wetland is a result of the combination of hillslope hydrology and channel hydrology, as well as small-scale dynamics within the wetland area. Hydrologic inputs of precipitation and runoff flow laterally over topography and vertically through substrates carrying solutes that are transported through different pathways (Burt and Pinay, 2005). Studies linking hydrology and transport of solutes have reported different hypotheses of how pore-storage acts under various conditions. Raisin et al. (1999) found that groundwater exchange accounted for 97% of the surface storage in small natural wetlands, and attributed up to 50% of the TN and TP load leaving the wetland through flushing of pore water. Yavitt et al. (2006) investigated solute transport of an in-stream wetland through analysis of soil organic matter and suggested that the wetland behaved like gelatin by holding water tightly as stream water flowed over the wetland matrix. Together, these findings suggest that flow pathways and chemical pathways are complex and may be very site specific by nature. Variations in water table and resultant gradients in storage recharge to streams make it necessary to investigate biogeochemical fluxes in periods that are classified based on hydrologic conditions, meaning periods of recharge and discharge (Rucker and Schrautzer, 2010).

Evapotranspiration in a Florida, USA, freshwater wetland was found to be as high as 1.00 cm/d, a range of 0.12 mg/d in February and 0.56 cm/d in May, and a total annual loss of 131.67 cm (Dolan et al., 1984). This study showed the magnitude of temporal variability and the significance of this water budget component on annual trends. In a comparison study, Sumner and Jacobs (2005) examined the utility of the common ET estimation methods of Penman-Monteith, Priestley-Taylor, reference evapotranspiration, and pan evaporation methods in predicting measured ET losses (Sumner and Jacobs, 2005). The authors found that the Priestley-Taylor method provided the best relation with measured ET when the dimensionless correction factor (alpha) was empirically determined as a function of green-leaf area index and solar radiation. In a riparian wetland setting, the most significant aspect of a riparian wetland setting with respect to ET is that they are directly adjacent to stream channels and that the characteristics of the cover changes between open water, varying vegetation density, and bare soil along a long and then shaped wetland (Drexler et al., 2004).

Recent studies on the extent and scale of floodplain-groundwater and surface water exchange emphasize the need to expand the scale of surface water nutrient fate and transport from the traditional focus on in-channel processes to encompass the entire active floodplain (Woessner, 2000). Complex interactions of adjacent topography with stream channels affect riparian hydrology (Claxton et al., 2003; Winter, 1999). Due to the complex nature of groundwater and the period of time needed to characterize water-table fluctuations, few wetland studies have completely characterized the hydrologic budget to include groundwater exchange, despite its potential role in nutrient fate and transport (Bradley and Gilvear, 2000; Raisin et al., 1999).

Wetland Hydraulics

Design recommendations for constructed stormwater wetlands call for the integration of different macrotopographic features into the design to control settling of suspended solids in specific areas, introduce flow complexity to slow moving water, and provide varying hydrologic patterns for a heterogeneous standing crop of emergent vegetation (ART, 1997; Schueler, 1992; VADCR, 2010). These macrotopographic features along with wetland vegetation create spatial variability in hydraulic parameters that govern flow through wetlands.

Hydraulic transport processes in constructed wetlands influence the residence time of flow through these systems, which affects the fate and transport of solutes (Keefe et al., 2004). Modeling of constructed wetlands as non-ideal flow through reactors has gone beyond the limitation of the assumptions necessary to describe wetlands as either plug-flow reactors (PFRs) or fully-mixed reactors (or continuously stirred tank reactors, CSTRs) to incorporate axial dispersion and mixing (Kadlec, 1994). Wetland flow modeling has been used to check for short-circuiting (Crohn et al., 2005; Martinez and Wise, 2003) and to determine pollutant removal rate coefficients, which are impacted by the effective volume ratio of a wetland (Persson and Wittgren, 2003).

Hydraulic parameters are coupled with information about loading rates in inflow concentrations of pollutants to inform design of treatment wetlands (Kadlec, 2000). The challenge of hydraulic parameterization of a constructed wetland for performance prediction lies in capturing the spatial variability mechanisms that drive pollutant removal. Tracer injection experiments provide the data necessary to quantify a variety of these hydraulic processes. Important factors to consider in performing an efficient tracer experiment include adequate flushing of previous tracer, use of inert tracer, adequate sampling frequency, and accurate knowledge of wetland water volume (Werner and Kadlec, 2000).

Biogeochemical Cycling

Riparian wetlands play an important role in nutrient transport and cycling (Lowrance et al., 1985; Seitzinger, 1994; Vellidis et al., 2003). The cyclic pattern of connection and disconnection with main channel flow can be quantified by the inter-flood interval (IFI), and rates of metabolic and biogeochemical cycling of materials have been attributed to the hydrologic control of the IFI of riparian areas (Valett et al., 2005). The flora and fauna that inhabit riparian areas are adapted to the successional patterns that are created by this dynamic hydrologic regime (Naiman and Decamps, 1997).

Abiotic and biotic factors affect the fate and transformation of nutrients within a wetland. Temporal variations in temperature, solar energy, and hydrologic inputs change delivery pathways and sequestration processes (Ward and Stanford, 1995). Steep redox gradients are also characteristic of the interfaces between saturated soil and floodwater and between antecedent wetland storage water and stormwater inflows. At these interfaces, there exist micro gradients of pH, dissolved oxygen, and concentrations of other electron acceptors for microbial processes (Reddy and DeLuane, 2008). Reddy and D'Angelo (1997) suggested biogeochemical indicators within constructed wetlands can be used to evaluate wetland nutrient processing (Table 1.3).

Table I.3
Summary of biogeochemical indicators of pollutant removal efficiency in constructed wetlands (after (Reddy and DAngelo, 1997)).

Pollutant of Concern	Biogeochemical Indicators
Carbon, toxic organic compounds	Soil or water oxygen demand Microbial biomass
	Soil Eh
	Soil pH
Nitrate	Dissolved organic C
	Microbial biomass
Phosphorus	Reactive available Fe and Al
	(acidic soils)
	Reactive available Ca and Mg
	(alkaline soils)

Sediment – There exists a relationship between sediment that is moved within the water column (total suspended solids, or TSS) and hydrologic factors including the amount, intensity, and duration of precipitation, as well as water flow rates ((Reinelt and Horner, 1995). Sediment trapping and redistribution within wetland topography is a function of flow velocity and the existing topography itself (Noe et al., 2010). Studies of the topography of the Everglades concluded that sediment redistribution along with peat accretion processes in wetlands were crucial to perpetuating the ridge and slough topography of the system (Larsen et al., 2007).

Carbon - The carbon (C) cycle within wetlands is mainly temporary storage of organic matter and decomposition or decay. Temporary C sinks in a wetland include plant biomass, detritus, microbial biomass, and dissolved gases. Different types of bacteria break down complex organic materials through enzymatic hydrolysis, resulting in a variety of simpler C compounds (Reddy and DeLuane, 2008). These compounds are then mobile and available for further reduction ultimately to carbon dioxide under aerobic conditions or methane under anoxic conditions (more commonly found in wetlands) (D'angelo and Reddy, 1999).

Nitrogen - The nitrogen (N) cycle within wetlands is characterized by the transformations between nitrogen species that are partitioned into the water column, substrates, and biomass. Physical processes that govern these transformations include atmospheric deposition of nitrogen, ammonia sorption, ammonia volatilization, and sedimentation of particulate N. Dominant microbial processes in the wetland N cycle include ammonification of organic nitrogen, nitrification of ammonia (an aerobic process), facultative heterotrophic denitrification (mainly an anaerobic process), and N fixation by soil bacteria. Other processes include mineralization as soil organic compounds break down over time and immobilization of ammonium-N in microbial biomass. Vegetation affects the wetland N cycle through pulsing effects of uptake of nitrate and ammonia during the growing season and release of ammonium-N during decay. Fractions of N inputs into a wetland become permanently stored within the wetland soil substrates as burial occurs over time (Kadlec, 2009a).

Temporal variations of N removal from source waters exist in constructed wetland systems. For example, Sartoris et al. (1999) found that the ammonia-N demand from newly planted wetland vegetation was the dominant removal mechanism within the first two years after construction. After vegetation grow-out, the fraction of open water surface area decreased, resulting in limited re-aeration and subsequent limited nitrification. After a reconfiguration of the study wetland used by Satoris et al. (1999) where more deep water open area were introduced, nitrification within the wetland was enhanced (Smith et al., 2000).

The storage of N within a FWS wetland is most abundant in the subsurface in wetland soil substrates. A significant portion of N is also stored in standing crops of vegetation (living and dead). The least amount of N is stored in the water column in mobile forms (Kadlec, 2009a). N removal and release has been attributed to the displacement of antecedent wetland storage water (Kadlec, 2010), denitrification potentials mediated by dissolved oxygen levels and temperature (Rucker and Schrautzer, 2010), and seasonal variability in inflow concentrations and hydraulic residence time (Woltemade and Woodward, 2008).

Phosphorus - Wetland phosphorus cycling can be compartmentalized into P in the water, plants, microbiota, organic litter and humus, and soil substrates (Kadlec, 2009a). P storage within and transport between these compartments is a dynamic equilibrium that is controlled by chemical P concentrations, abundance of potential mineral precipitants, microbial activity, and redox conditions (Reddy et al., 1999). Long-term storage of P is a function of accretion rate of soil substrate within the wetland and sorption potential to those substrates (Aldous et al., 2007).

P is also removed from the water column through periphyton-mediated processes. Periphyton uptake and deposit P, filter particulate P, and decrease advective transport of P in the water column by attenuating flow and the particulate and dissolved P that is transported with it. Periphyton also create local increased pH to the point that the precipitation of calcium phosphate and carbonate-phosphate complexes may occur, resulting in long-term burial of P within wetland substrates (Dodds, 2003).

P sorption chemistry is very complex, but there are general trends in phosphorus activity. In acidic conditions, P may be fixed by available aluminum, manganese, and iron oxides. In alkaline conditions, P may be fixed by available calcium and magnesium. Under reducing conditions, the solubilization of iron minerals leads to the release of fixed P (Reddy and D'Angelo, 1994).

P enters a wetland through various transport mechanisms that are controlled by site-specific topographic characteristics. Hydrologic pathways for phosphorus delivery include overland flow, hyporheic flow, flooding inundation, drainage flow, diffuse flow from adjacent hillslopes, and direct flow from deep recharge pathways (Hoffmann et al., 2009). Each of these delivery mechanisms has an effect on the abundance and form of phosphorus in a wetland.

P is of particular concern because it is often the limiting nutrient for primary productivity and therefore controls the relative amount of standing crops of aquatic autotrophs. These autotrophs are the primary control of dissolved oxygen levels in surface waters, which is a sensitive water quality parameter for fish and macroinvertebrates. Currently, there are no surface water standards, but the USEPA (1986) recommends below 0.50 mg/L TP for surface waters. However, levels of TP in excess of 0.03 mg/L TP have been determined to produce nuisance algal blooms in temperate lotic environments (Dodds, 2002).

In a comprehensive review of literature on wetland nutrient removal, riparian wetlands were found to reduce TP loadings (Fisher and Acreman, 2004). However, loadings of soluble P were likely to increase rather than decrease. Greater TP concentrations in ground water associated with poorly drained riparian buffers suggested that wetland designs for nitrate removal might not be effective for P removal (Young and Briggs, 2008). However, the highly dynamic hydrology associated with riparian areas combined with native wetland vegetation may create favorable conditions for capture of particulate and dissolved P that may be traveling through preferential groundwater flow-paths or in overland stormflows (Braskerud, 2002b; Fisher et al., 2009; Van de Moortel et al., 2009; Wetzel, 2001).

P storage in a FWS wetland is primarily within the subsurface soil root zone. Some P storage occurs in standing vegetation. Relatively small amounts of P are stored in the water column as mobile forms (Kadlec, 2009a).

I.4 Summary

The body of knowledge on wetland ecological functions is both diverse and deep. The hydrologic and nutrient processes that occur within wetlands have been identified as those that are desirable for preservation and re-creation. BMPs for the enhancement of ecological service to mimic those provided by wetlands have become common tools for water quality management. Despite our depth of knowledge of wetland processes, we still lack basic understanding of optimal creation practices in various landscape positions. This is apparent through the abundance of failed wetland creation or restoration sites and stormwater management practices throughout the country.

To gain the tools necessary to inform better design of created wetlands and BMPs, we need to continue studies on the effectiveness of the state-of-the-science design recommendations. Here, we design, implement, monitor, and evaluate an innovative stormwater BMP. The innovation is in the landscape placement of the BMP – the floodplain – and the design was created through the implementation of concepts and guidance provided by state agencies in technical manuals. The overall goal of this research is to evaluate the effectiveness of an integrated practice for controlled NPS pollution on an event time-scale, which is consistent with the practice's HGM placement within a stream network.

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II. Methods

II.1 Study Area

Opequon Targeted Watershed

The USEPA and the Chesapeake Bay Program have identified watersheds for targeted efforts in stormwater management for the reduction of nutrients reaching the Chesapeake Bay. Opequon Creek is one of these targeted watersheds (Fig. 2.1). The headwaters of Opequon Creek are located in the Ridge and Valley physiographic province of Northern Virginia. Opequon Creek flows north into West Virginia before it ultimately reaches the Potomac River. A total maximum daily load (TMDL) was identified for the Virginia reach and addressed benthic and bacterial impairments. A TMDL is also identified for the West Virginia reach and identifies biological impairment (Mostaghimi, 2003a; Mostaghimi, 2003b). In 2006, the Biological Systems Engineering Department secured a grant from the USEPA and the National Fish and Wildlife Federation for water quality studies. Collaborative partnerships within the watershed associated with the grant provided the funding for the design, implementation, and monitoring of innovative stormwater best management practices.

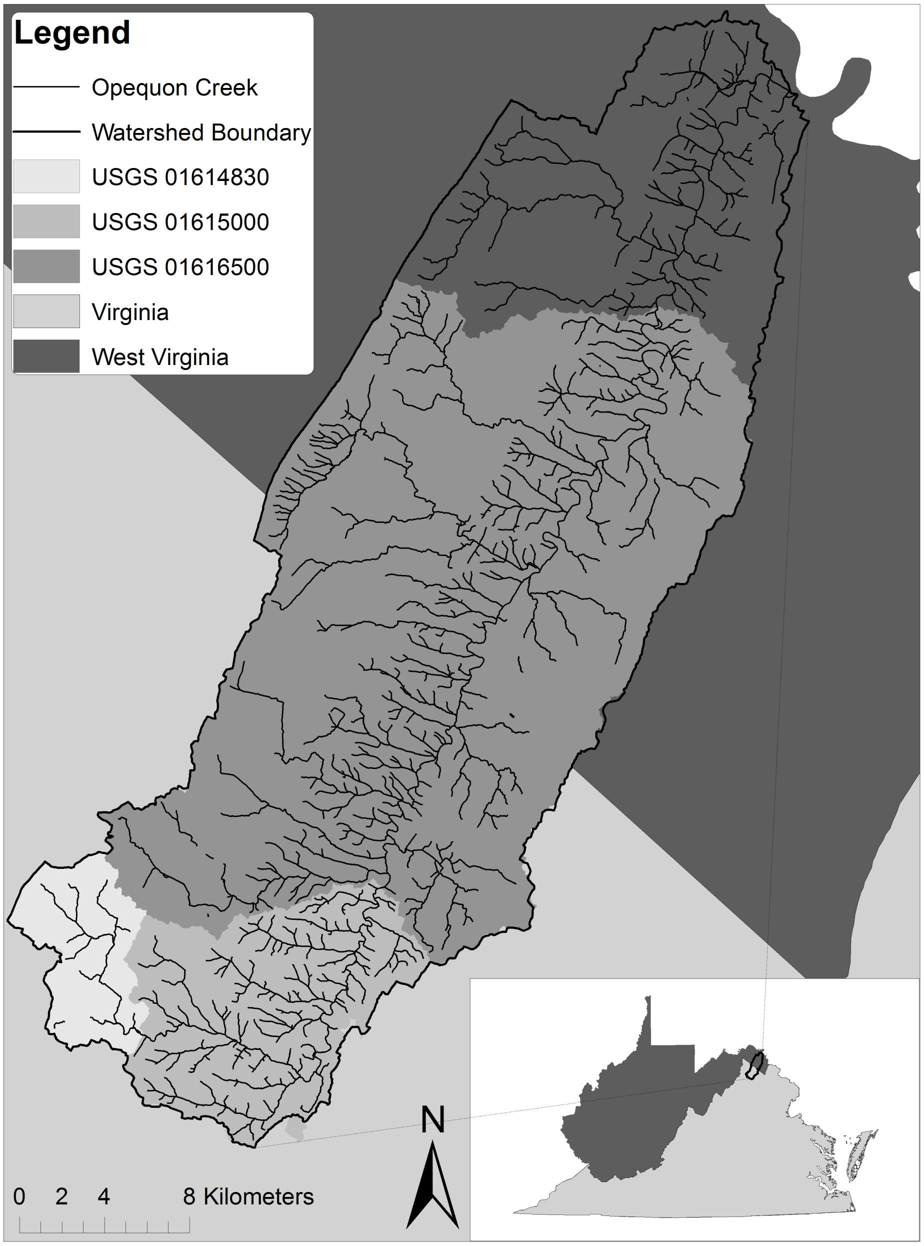

Figure II.1 Opequon Creek basin in northern Virginia and eastern West Virginia.

II.2 Wetland Implementation

Design

A 0.2-ha wetland was constructed in the floodplain of Opequon Creek just south of Winchester, VA, on Hedgebrook Farm, which recently celebrated its 100-th year of operation. The constructed wetland (CW) was designed as a riparian slope wetland to receive hydrologic inputs from overbank flows from Opequon Creek and groundwater exchange. CW design and implementation methods drew from Integrated Constructed Wetland (ICW;(Zedler, 2003) concepts by considering site-specific characteristics and aesthetics desires of the landowner in every step of the process.

The wetland was designed using constructed wetland guidance from the Virginia Department of Conservation and Recreation (VADCR, 1999), incorporating three depth zones throughout the wetland to increase complexity: deep pools, a low marsh slough, and high marsh areas. Depths and surface areas of these features were determined based on VADCR (1999) guidance and local conditions. The plan-view layout of the wetland was guided by a combination of site-specific characteristics, landowner input, and examples from the VADCR (Fig. 2.2). The low marsh meanders through the high marsh, creating variably inundated areas within the wetland, as well as two different flowpaths, one for low-flow conditions and another for high flows (Fig. 2.3).

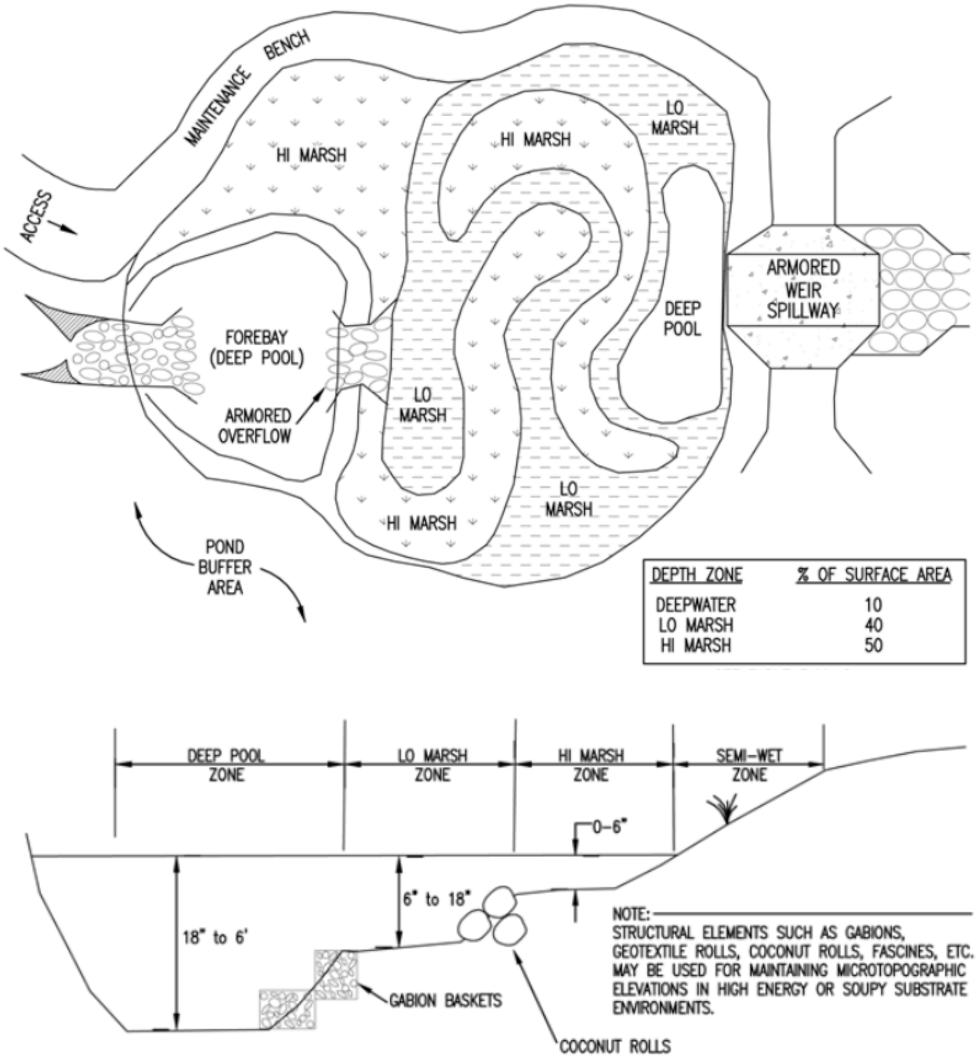

Figure II.2 Plan and profile schematic of a constructed stormwater wetland and associated dimensions of integrated features (VADCR, 1999).

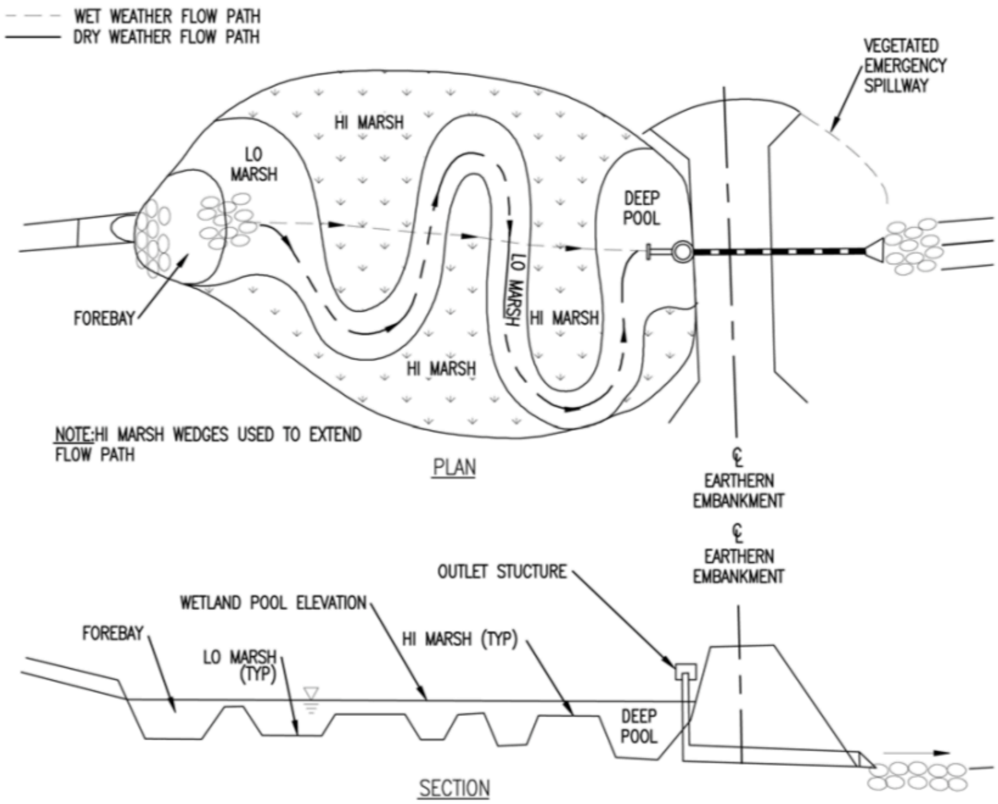

Figure II.3 Plan and profile schematic of dry weather and wet weather flow paths through a constructed stormwater wetland (VADCR, 1999).

We decided to design the CW to receive overbank flows from the channel at or above bankfull. In our design, bankfull elevation is the stage in the stream where flows would be clipped off and diverted through the inlet (NRCS, 2008). The bankfull channel was determined through field indicators (Leopold, 1962), regional curves (Keaton et al., 2005), and flow frequency analysis (Ries, 2002).

A cross-section of the channel at the wetland inlet was surveyed using a laser level, and a bench in the profile was an indicator of a bankfull channel (Leopold, 1962)(Fig. 2.4). This elevation was consistent with existing wetland rushes in the middle of the floodplain. Regional curve data for streams in the non-urban Ridge and Valley Physiographic Province of Virginia published by the USGS related a watershed of approximately 40 km^2 to a 1.5-year discharge of 11.3 cms (regression data range of 2.8-22.6 cms) (Keaton et al., 2005).

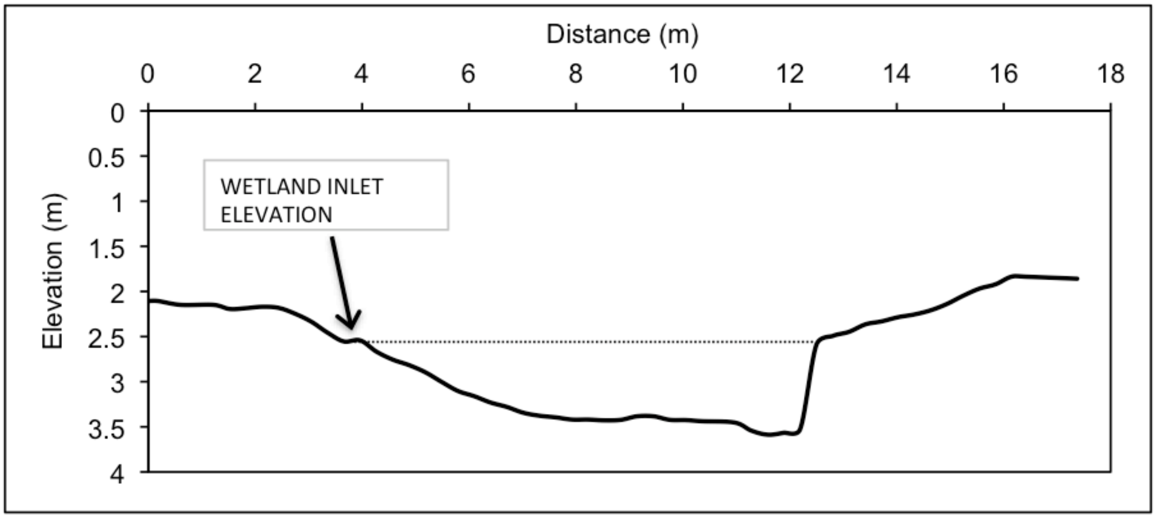

Figure II.4 Wetland inlet elevation as relates to the Opequon Creek channel cross-section at the location in the bank where excavation occurred.

Bankfull flow (return period T=1.5 yrs) was also calculated with a Log Pearson Type II estimation using flow records from two USGS gages: 1) a 9-year flow record gage station located approximately a mile downstream (USGS Gage 01614830, drainage area = 39.4 km^2; Figs. 2.5, 2.6) and 2) an area-weighted estimation using 61-year period of record further down Opequon Creek (USGS Gage 01615000, drainage area= 150.7 km^2). These calculations resulted in a bankfull flow of 0.6 cms (Fig. 2.7) and 8.3 cms (2) (Fig. 2.8), respectively. The discrepancy between the two may be attributed to the significant difference in length of record and the effects of a multiple-year drought that occurred in the region during the 9-yr short record. In the final design, macrotopographic feature elevations were selected based on the field evidence of bankfull and mapped wetland vegetation.

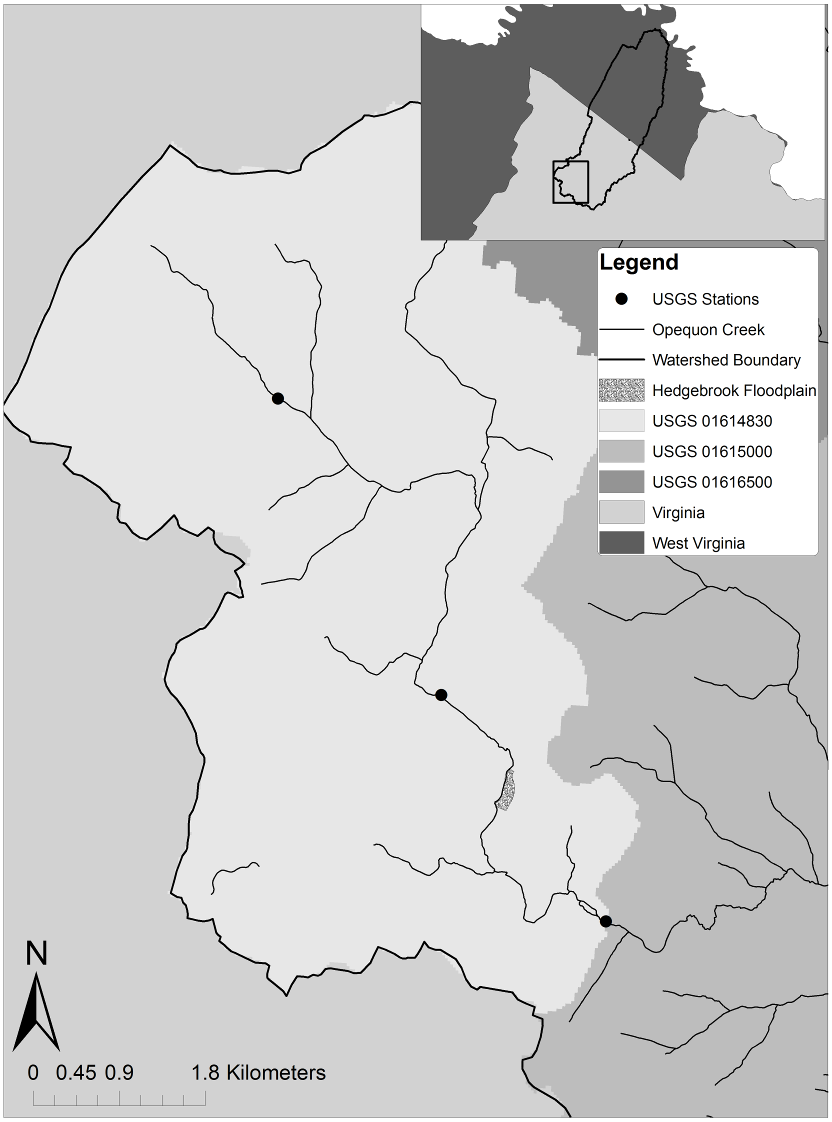

Figure II.5 Opequon Creek basin delineated from location of United States Geological Survey gage station 0161483, near Stephens City, VA.

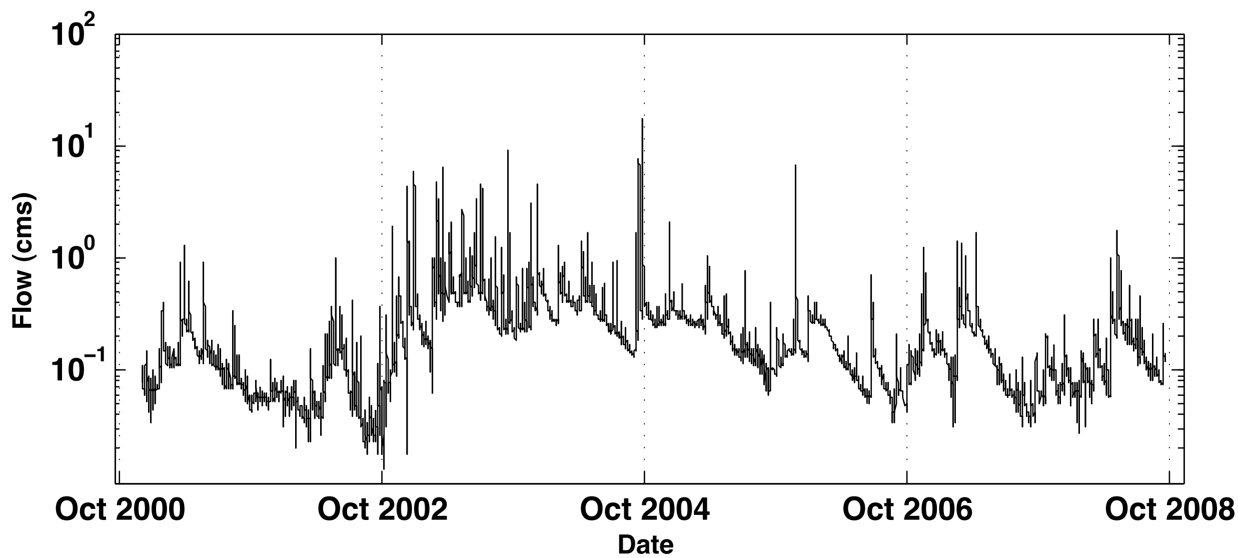

Figure II.6 Flow record from USGS Gage 01614830 on Opequon Creek, just outside of Stephens City, VA.

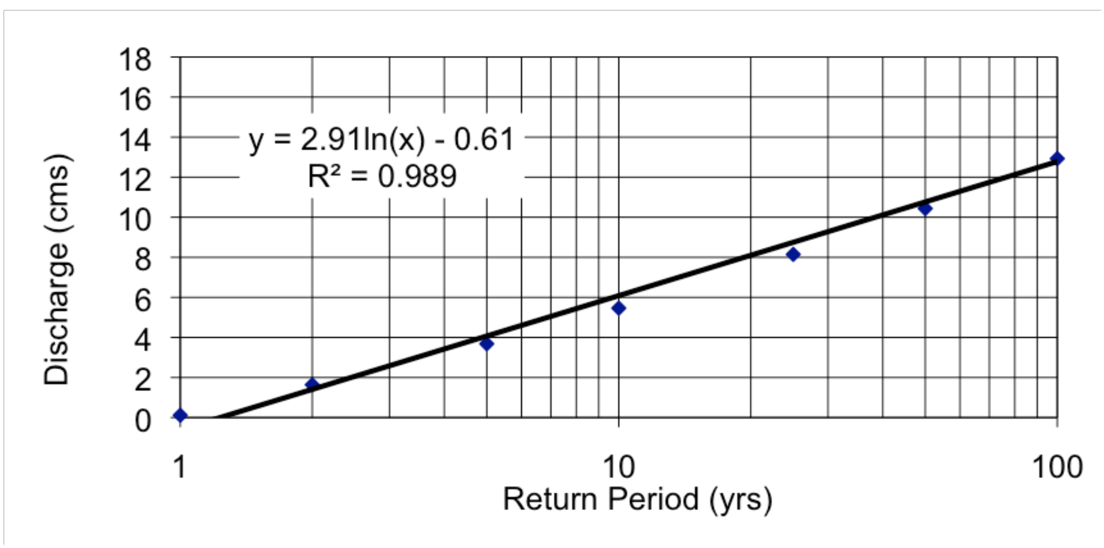

Figure II.7 Return frequency discharges (cms) for Opequon Creek calculated from daily flow data from the United States Geological Survey Gage 01614830 near Stephens City, VA (drainage area = 40 km^2).

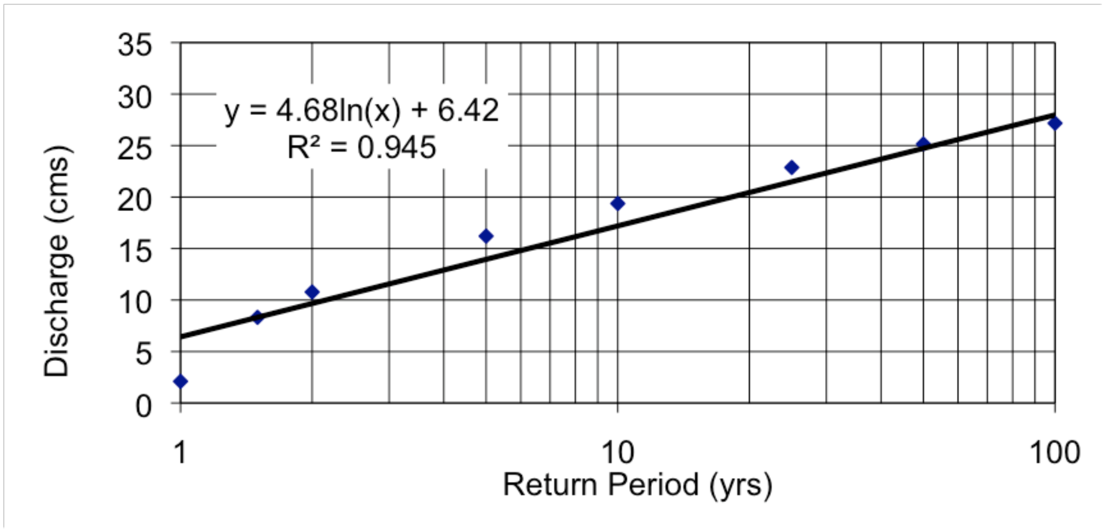

Figure II.8 Area-weighted return frequency discharges (cms) for Opequon Creek calculated from daily flow data from United States Geological Survey Gage 01615000 outside Berryville, VA (watershed area ratio: 0.26).

The inlet and outlet channels were cut into the creek banks once an annual rye cover crop had established on site in July 2007. The inlet and outlet channels are earthen grass channels that were sized to carry overbank flows. The bank angles of the channels are 3:1 sloping banks designed to minimize erosion and scour (VADCR, 1999). The elevation of the inlet channel bed was selected to match that of the determined bankfull channel stage and design high marsh elevation in order to allow access of floodwaters to the floodplain while not disrupting the dynamic equilibrium that exists between stable channel geomorphology, sediment transport, and floodplain interactions with the creek channel (NRCS, 2008).

The wetland surface area is approximately 2040 m^2 (0.2 ha) and surface-water storage volume is 225 m^3 at the capacity of the outlet flume (Table 2.1). The general slope of the wetland bed is 0.5%, with average dimensions of 72 m by 25 m; however, the preferential flowpath of the wetland through the pools and low marsh create a flowpath of 85 m and a length:width ratio of just over 3:1 (Fig. 2.9). At capacity, the depths in the wetland vary from 0.15 m, 0.40 m, and 1.0 m in the high marsh, low marsh, and pools, respectively.

Table II.1
Wetland dimensions as measured by post-construction topographic surveying.

Wetland Dimensions		
Feature	**Area (m^2)**	**Depth (m)**
Pools	240	1.00
Low marsh	355	0.40
High marsh	845	0.15
Berm side slope	600	na
Total	2040	na

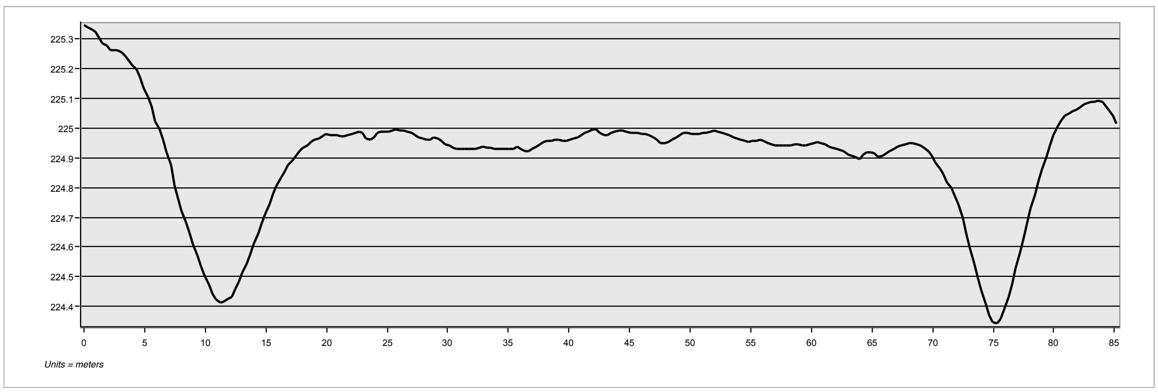

Figure II.9 Longitudinal profile of wetland flowpath through macrotopographic features.

Construction

At construction, the first challenge was to address the existing fescue mats that dominated the floodplain pasture. Since Hedgebrook Farm is a pesticide-free operation and proximate to a highway, spraying or burning the area was not an option as a removal method. The aggressive grass *Festuca glauca* (fescue) and the associated seedbank stored in the top layer of soil were manually removed by an initial scraping of the top 10 cm. Wetland macrotopographic features were flagged into the site, and approximately 650 m^3 of floodplain soils were excavated. During excavation, care was given to minimize compaction as much as possible. The area where existing palustrine wetland vegetation had been mapped was handled with particular care, and only one pass was taken over that area to remove the fescue. The entire constructed wetland area was seeded with an annual rye and wetland grass mix, and a final till was performed.

The vegetation-planting plan was guided by elevation and anticipated standing water depth in the macrotopographic features (Fig. 2.10). The Northern Virginia Chapter of Master Gardeners provided expertise to guide species selection and placement as well as volunteer labor for planting. The low marsh was planted with *Sagittaria latifolia* (arrowhead) and *Pontederia cordata* (pickerelweed), while the high marsh was planted with *Scirpus validus* (bulrush), *Acorus calamus* (sweetflag), and an additional wetland grass seed mix (Table 2.2). A timeline of events occurring at the site shows the milestones in wetland implementation and monitoring (Fig. 2.11). The photographs in Fig. 2.12 depict the site before and after wetland implementation. A photograph documentary of the project history can be found in the Appendices.

Table II.2
Planted wetland vegetation within constructed floodplain wetland.

Common Name	Scientific Name	Feature Placement	Rooting Depth (cm)
Arrowhead	*Sagittaria latifolia*	Low Marsh	45
Bulrush	*Scirpus validus*	High/Low Marsh	30
Pickerelweed	*Pontederia cordata*	Low Marsh	25
Sweetflag	*Acorus calamus*	High Marsh	25

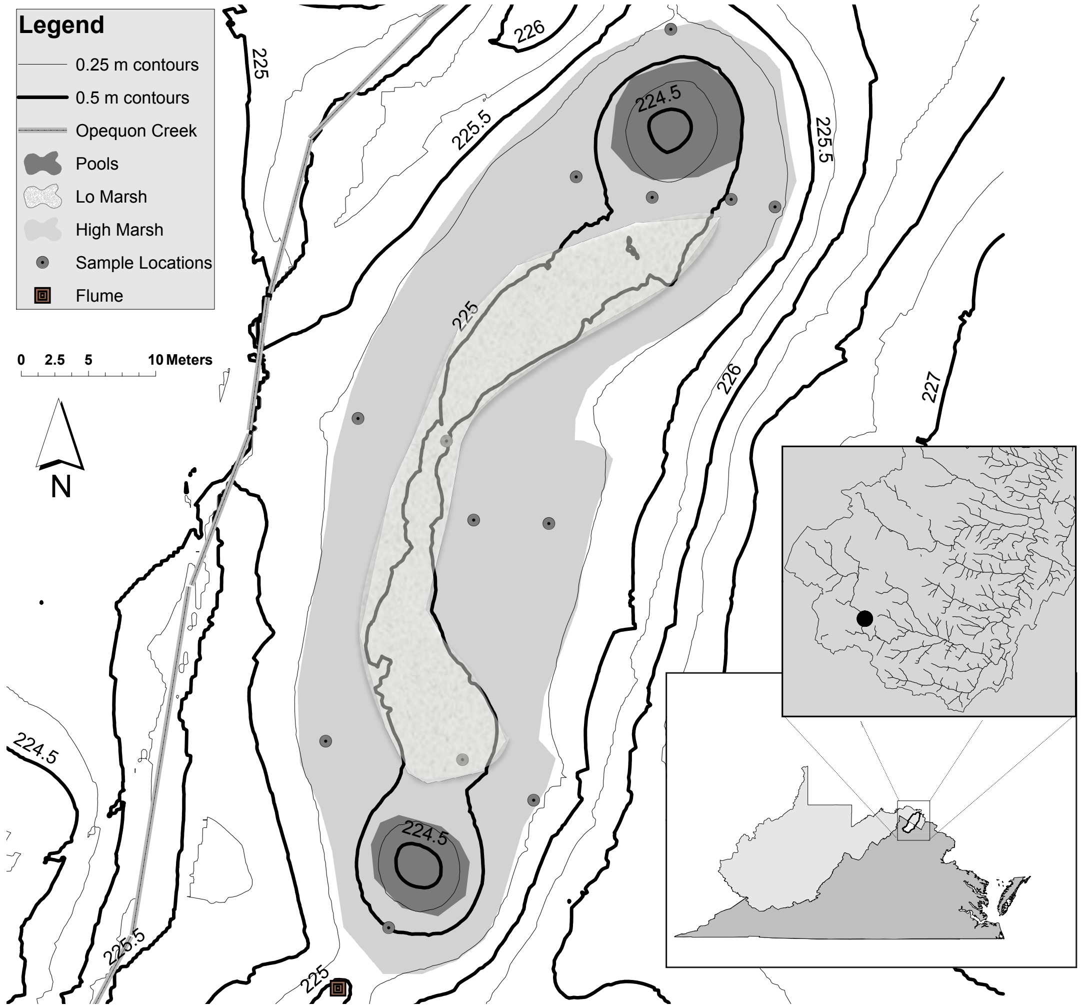

Figure II.10 Schematic of wetland topography, including design features of pools and marshes. Located in the Opequon Creek Basin in Northern Virginia.

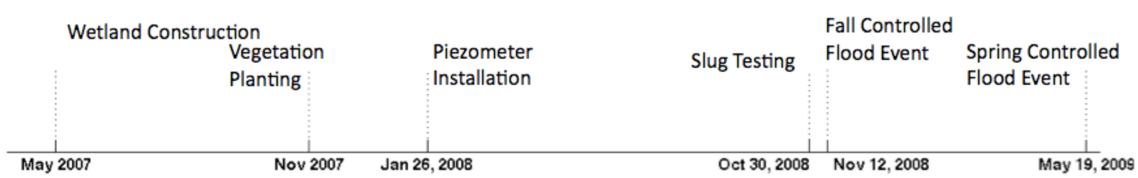

Figure II.11 Timeline of milestones for the implementation and monitoring at Hedgebrook Farm constructed floodplain wetland.

Figure II.12 Photographs of floodplain site at Hedgebrook Farm before constructed wetland implementation (left) and after implementation (right).

II.3 Groundwater Hydrology

Piezometers

Five nests of groundwater piezometers were installed in January 2008. Piezometers were constructed of 3.81-cm diameter solid PVC of various lengths (length required to reach datum plus a significant riser to extend through tall grasses), a 15-cm long slotted PVC section (0.025-cm slot thickness) that was coupled to the riser, and a flush-fitting PVC drive point at the end. Water-level measurements were recorded at each piezometer datum, which is the elevation of the 10-cm long slotted pipe portion below the riser (approximately 5 cm of the slots were closed due to glue from coupling). Piezometers were installed in the wetland by coring to the desired depth with an 8.9-cm hand auger, inserting the piezometer, backfilling with coarse sand to a depth above the slot, backfilling to within 15 cm from the surface with the cored soil, and then capping with bentonite to provide a water-tight seal around the PVC riser to eliminate preferential flow to the subsurface (Fig. 2.13)(USACE, 2000). Piezometers were then developed by the addition of a slug of water to flush out small particulates.

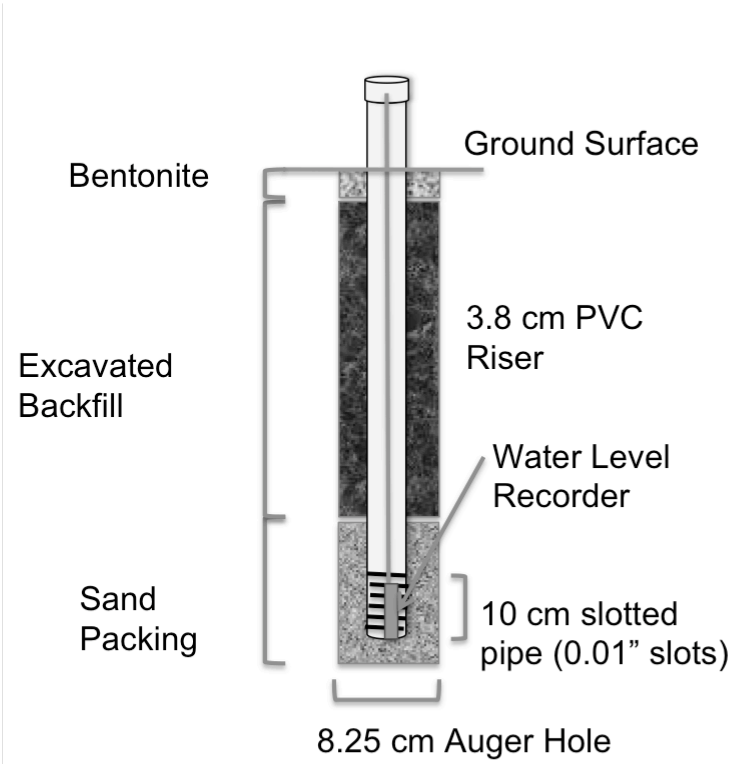

Figure II.13 Schematic of deployed piezometer in soil profile and associated datum for water level records using a continuously monitoring pressure transducer.

Locations of piezometer nests were determined by selecting a central location in the high marsh and using triangulation over the wetland surface area, capturing perpendicular and parallel flows from hillslope across floodplain to stream, and along the hydraulic gradient down the floodplain (Fig. 2.14).

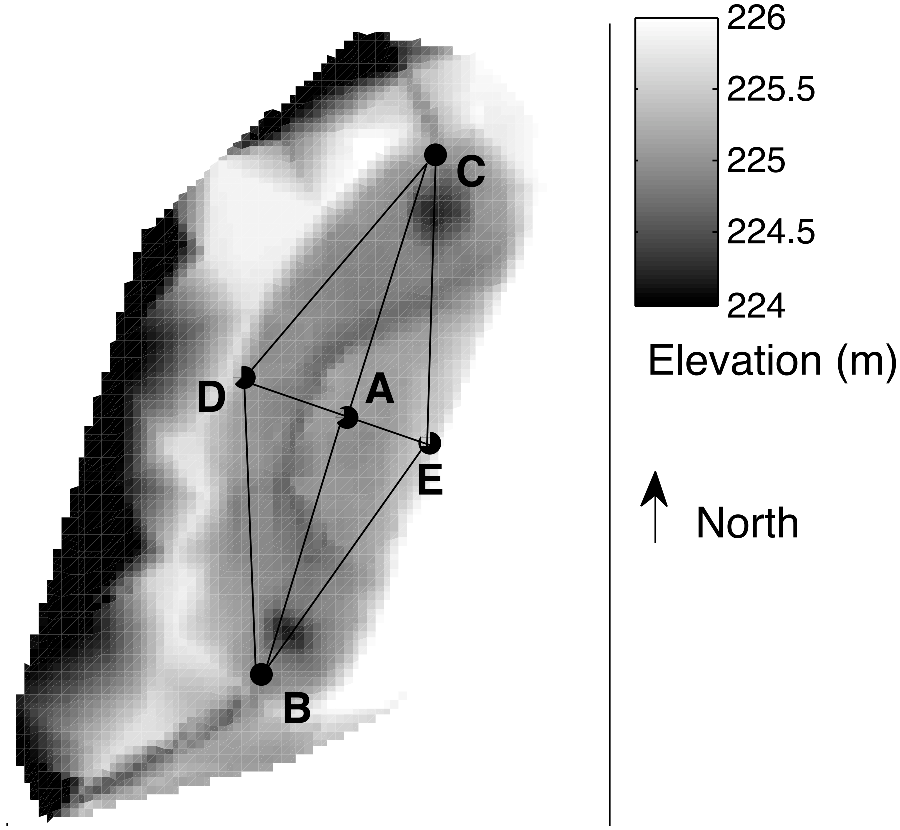

Figure II.14 Triangulation pattern used to interpolate hydraulic gradients in planes between piezometer data points.

Each piezometer nest contained three piezometers; a deep piezometer that extends below a characteristic clay layer, a middle piezometer that is located in a thick clay layer, and a shallow piezometer that is above the clay layer. Individual piezometer datum were selected by field observations during soil coring based on changes in the soil profile layer color and texture (refer to Chapter 3 for detailed figures). Absolute pressure and temperature measurements within each piezometer were continuously monitored with water-level loggers (Onset Corp., Bourne, MA). An additional probe was incorporated in October 2008 to record atmospheric pressure in a mock piezometer casing in the wetland berm. Previous to this, atmospheric pressure was measured in dry piezometers and at a proximate weather station (6.4 km away) and the atmospheric pressure at the wetland site was determined through a relationship between the two when there were no dry piezometers.

II.4 Mapping

Wetland topography was captured with a Topcon total station (Topcon Inc., Tokyo, Japan) in August 2008. Three permanent benchmarks were installed in large trees and the survey was relative to elevations given to these benchmarks. Additional surveying was performed after the installation of the inlet and outlet channels. Piezometers were surveyed after their installation and aligned with the initial post-construction surveys using the Translate and Rotate Tools in the Foresight software package (TDS Inc., Westminster, Colorado, USA). In October 2009, benchmarks and piezometers were shot using a GR3 Survey-Grade GPS Unit (Topcon Inc, Tokyo, Japan) along with a new permanent benchmark, whose location was satellite-identified down to millimeter resolution. All the previously aligned surveys were then aligned to the new permanent benchmark. Through comparisons of multiple benchmark shots, the associated error with the alignments of various surveys was estimated to be 10 cm. Since the piezometers were shot with the GR3, the error in their elevations relative to each other was less than 1 cm.

II.5 Bulk Density

On October 30, 2008, bulk density samples were taken at nine locations throughout the wetland; six locations in the high marsh and three locations in the low marsh. Samples were collected using a standard sampling hammer and 5-cm diameter sampling rings. Samples were collected from within the top 7 cm of soil and wrapped in plastic for transportation. Samples were dried overnight in an oven at 105°C and then weighed on a calibrated scale (ASTM C29). Bulk density was determined as the mass of dried soil per volume of soil sample (Appendix A).

II.6 Soil Sampling

On November 11, 2008, soil samples were collected at 12 sampling locations throughout the wetland; nine locations in the high marsh and three locations in the low marsh. Samples were collected using a 10 cm long hand trowel and stored in plastic lock bags. Each sample was composed of two subsamples: soil from the top 5 cm, and soil from the underlying 10 cm. Each subsample was sent to the Virginia Tech Soil Testing Lab for the standard analysis of acid-extractable ions, pH, cation exchange capacity, and organic matter as determined by loss on ignition (Appendices B and C).

II.7 Wetland Hydraulics and Nutrient Assimilative Capacity

Artificial Overbank Events (AOEs)

Artificial overbank events were created at the wetland site to simulate inundation and flow-through conditions; one performed in the fall (November 12, 2008) and another in the spring (May 19, 2009). These controlled events provided ideal conditions to answer research questions in the realm of hydraulics and nutrient assimilative capacity (Chapters 4 and 5 describe specific research objectives).

The AOEs were created by pumping Opequon Creek water into the wetland with a John Deere 76-hp trash pump (John Deere, Moline, Illinois, USA). Each AOE was designed to last approximately 8 hours and to produce a 4.25 m^3/min inflow rate. These conditions were selected based on the goal of trying to produce an event that would mimic the duration and intensity of an overbank flow estimated from flow records at USGS Gage 01614830 while suiting the practical needs of the data collection experiment. A conceptual diagram of the produced hydrograph and the related sampling was created to inform activities in the field (Fig. 2.15). The inflow was amended with concentrated sodium nitrate and sodium phosphate slurry to attain nutrient concentrations that would mimic a natural storm event.

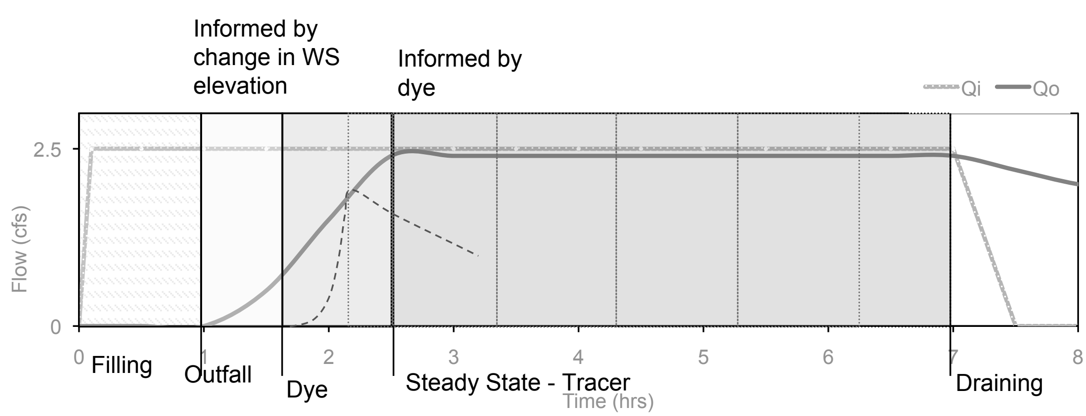

Figure II.15 Artificial overbank event conceptual hydrograph and associated sampling.

The wetland inlet and outlet were instrumented with 6712 Portable Samplers (ISCOS; Teledyne ISCO, Inc., Lincoln, NE). The inlet unit comprised a 750 Area Velocity Flow Module and automated sampling system, while the outlet unit utilized a 730 Bubbler Flow Module to capture stage in the 0.3-m H-flume and an automated sampling system. Inflow was quantified using continuity of flow through a known cross-section at a measured velocity, while a flume-rating curve quantified outflow based on stage measurements. The hydrographs of both AOEs indicate that a slightly higher average flow rate was attained in the spring event, and that in both events, the wetland drained within an hour after the pumping had ended (Fig. 2.16).

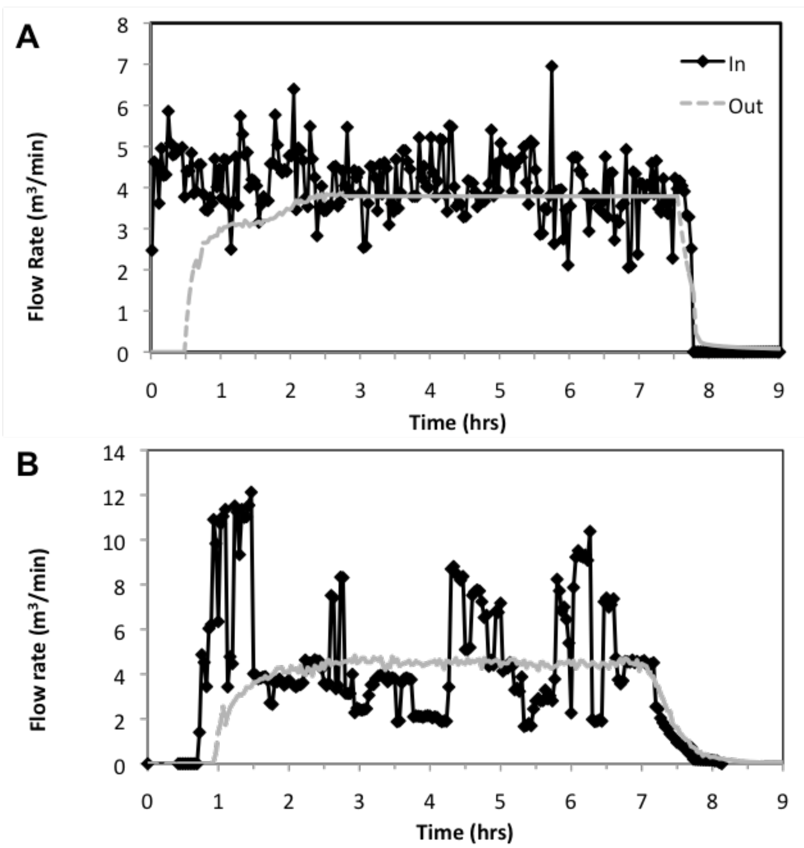

Figure II.16 Generated hydrographs from controlled pumping events in Fall 2008 (top) and Spring 2009 (bottom).

Four main sampling protocols were used to characterize the AOEs (Table 2.3):

- In/Out Sampling – These samples were used in percent mass removal calculations. ISCOS were programmed to collect composite samples based on elapsed time. The ISCO units malfunctioned at the beginning of the spring event, making them unreliable for the inlet-outlet sampling. Instead, grab samples in the inlet and outlet were used in place of ISCO composite samples for the spring event. The samples were filtered upon return to lab and stored at 4°C.
- Snapshot-Grab Sampling – These samples were used to identify variability in nutrient concentrations. Grab samples were collected simultaneously from the 12 sample locations (replicated four times during hydrograph plateau), filtered in the field, and stored at 4°C.
- Tracer Sampling – These samples were used to create breakthrough curves of the conservative tracer (bromide, Br) that quantified residence time distributions. Rapid simultaneous sampling occurred at pre-determined time intervals along the flowpath, and the samples were filtered upon analysis.
- Synoptic Sampling – These samples were used to characterize fine-scale spatial variability in pollutant concentrations. Grab samples were collected at 25 sample locations (including the 12 snapshot sample locations) within a 20-min window during plateau, filtered in the field, and stored at 4°C.

Table II.3
Field sampling protocols and techniques for controlled flood events to capture hydraulic tracer and nutrient data.

Sampling Protocol	Technique	Time Interval	Constituents of Analysis
In/Out	Automated ISCO	Time dependent Inlet – 20-30 min Outlet – 10-60 min	TSS, TN, TP, NO_3, NH_3, PO_4,Br
Grab	Field rinse bottle two times, grab from reachable water column excluding large floating particulates	Approximately 1 hr apart during steady-stage plateau	TSS, TN, TP, NO_3, NH_3, PO_4,Br

Sampling Protocol	Technique	Time Interval	Constituents of Analysis
Tracer	Rapid grab sampling	Location specific 3-20 min	Br
Synoptic (Spring only)	Field rinse bottle two times, grab from reachable water column excluding large floating particulates	Within a 20-min window during plateau	TSS, TN, TP, NO_3, NH_3, PO_4,Br
First Flush	Field rinse bottle two times, fill from outfall of H-flume	First 30 min of outfall	TSS, TN, TP, NO_3, NH_3, PO_4
Stream Walk	Field rinse bottle two times, grab from stream water column excluding large floating particulates	Within a 15-min window during plateau	TSS, TN, TP, NO_3, NH_3, PO_4

Twenty-five staff gages were installed throughout the wetland and monitored throughout the AOEs. The field identification of steady-state was determined as the time when wetland stage was steady. In the fall event, a florescence dye slug was added at the wetland inlet after steady-state had been achieved. The dye was visually tracked through the wetland and the time of dye breakthrough at the outlet was used to inform the spacing of the snapshot grab samples as well as the start of tracer sampling for the downstream transects. A complete timetable of events of each AOE can be found in Appendices G and H.

Suspended transects were installed to create access to the wetland storage during the AOEs without having to disturb the substrates or vegetation. Sampling locations were selected along these transects to capture a spatial distribution of high and low marshes along the wetland flowpath (Fig. 2.17). Sample locations were flagged along each transect and used for the snapshot-grab and tracer sampling. Locations were consistent between seasons with the exception of the hillslope high marsh location on Transect B, which was shifted 3 m for the fall AOE due to lack of inundation. During the steady-state hydrograph plateau, the wetted perimeter was flagged and then surveyed using the Topcon surveying unit. Surveys were subsequently aligned to the comprehensive wetland survey, and inundation surfaces were created for each event (Fig. 2.18).

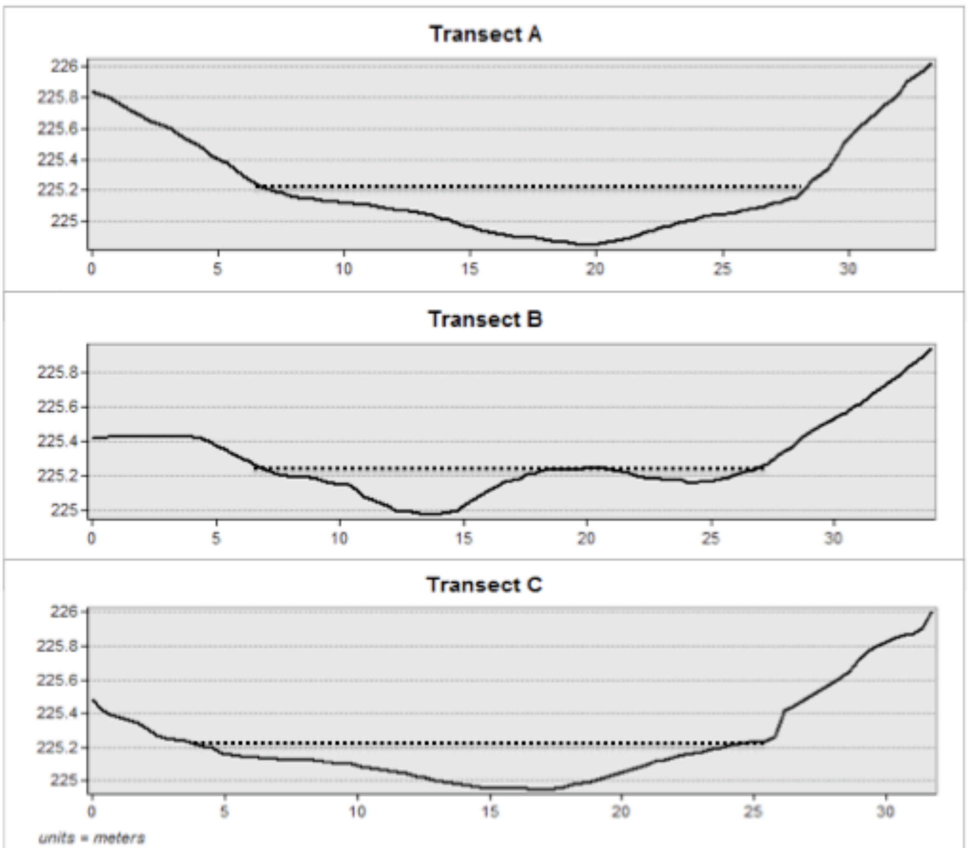

Figure II.17 Wetland topographic cross-section at each of the three sampling transects: Transect A just downstream of the inlet channel outfall and forebay, Transect B across the center of the wetland, and Transect C just upstream of the exit pool and outlet H-flume.

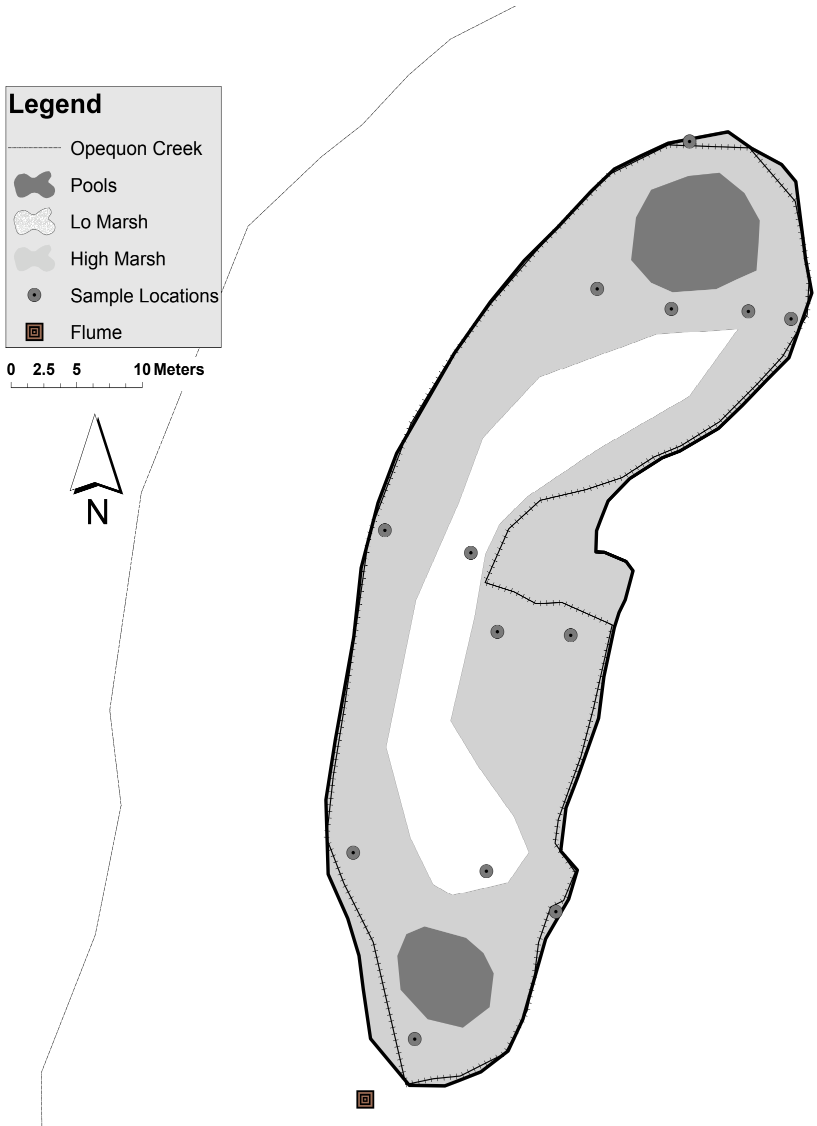

Figure II.18 Inundated wetland area during the Fall (November 2008) and Spring (May 2009) artificial overbank events.

II.8 Sample Handling and Laboratory Methods

New sample bottles were used whenever possible. Sample bottles that had been previously used were scrubbed with phosphate-free laboratory soap, acid rinsed in a 1 N HCl bath, rinsed three times with deionized water, and allowed to dry. A field rinsing method was employed during sampling to ensure any residue was rinsed before the water sample was collected. A field rinse consisted of partially filling the sample bottle with water, shaking to coat the bottle walls, dispensing the rinse water downstream, and repeating the process. Care was taken to not disturb benthic material during rinsing or sample collection.

When analyses called for field filtration, samples were filtered using a 0.45 μm filter attached to a geopump (GeoTech, Denver, Colorado, USA). Deionized water was pumped through the pump lines and filter between each sample. Samples were filtered and kept at 4°C in iced coolers when called for by the standard methods for analysis (APHA, 2000). Water samples for carbon analyses were acidified upon return to the lab with phosphoric acid (42% H_3PO_4). Analyses were performed according to established methods and standard methods for water analyses in the Department of Biology Stream Team Lab and the Biological Systems Engineering Department Water Quality Labs at Virginia Tech (Table 2.4). Field blanks were analyzed as unknowns with both sample sets collected during the controlled flood events. Laboratory blanks and replicates of 10% of samples were analyzed to ensure the quality of laboratory results. Analyses for NH_3 -N, NO_3 -N, and PO_4 -P, TN, and TP were completed using 4-point single-analyte calibration curves. Analyses for Br were completed using a 5- or 4-point single-analyte calibration curve. Instrumentation was properly calibrated before each analytical run. Ultra pure water was used in calibration standards and blanks.

Table II.4
Water quality analysis methods and pertinent information for the determination of targeted non-point source pollutants for stormwater and conservative tracer injection.

Constituent	Instrumentation	Method	APHA Method #	Operating Range
Total Suspended Solids (TSS)	Gravimetric Filtration Apparatus	Dried at 105°C	2540 D	>2 mg/L
Volatile Suspend Solids (VSS)	Muffle Furnace	Ignition at 550°C	2540 E	>0.2 mg/L
Ortho-Phosphorus (PO_4-P)	SEAL AA3 Autoanalyzer	Murphy & Riley colorimetric method with ascorbic acid reduction	4500-P G	2.5-100 μg/L P
Total Phosphorus (TP)	Autoclave, SEAL AA3 Autoanalyzer	Persulfate digestion	4500-P H	10-500 μg/L P
Nitrate+Nitrite (NO_3-N)	SEAL AA3 Autoanalyzer	Copper-Cadmium reduction	4500-NO_3 I	0.1-3.0 mg/L N
Ammonia (NH_3-N)	SEAL AA3 Autoanalyzer	Salicylate colorimetric reaction	4500-NH_3 H	2.5-100 μg/L N
Total Nitrogen (TN)	Autoclave, SEAL AA3 Autoanalyzer	Persulfate digestion	4500-N C	0.1-5.0 mg/L N
Bromide (Br)	Dionex 3000	Ion chromotrography	4110 B	0.5-15.0 mg/L Br
Non-Purgeable Organic Carbon (DOC)	Shimadzu TOC Analyzer	Acidification and Combustion	5310 D/B	0.1-10.0 mg/L NPOC
Total Dissolved Nitrogen	Shimadzu TOC Analyzer	Acidification and Combustion	5310 B	0.1-10.0 mg/L N

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III. Groundwater Hydrology of a Small Constructed Floodplain Wetland In the Ridge and Valley of Virginia

III.1 Introduction

Constructed wetlands (CWs) have been shown to provide eco-services that improve water quality and are often used as engineered best management practices (BMPs) for stormwater draining anthropogenically-altered landscapes (Guardo et al., 1995; Kadlec, 2009b; Mitsch et al., 2005). CWs have the potential to act as nutrient sinks, which are essential tools for nutrient management in ecologically sensitive areas, such as the Chesapeake Bay (Boesch et al., 2001) and in the face of changing climates(Seavy et al., 2009). The capacity of a CW to remove pollutants from stormwater is a function of site-specific physical and chemical characteristics of wetland substrates (Carleton et al., 2000; Kincanon and McAnally, 2004; Reddy et al., 1999) and the pollutant delivery mechanisms and hydrology of the area (Braskerud, 2002a; R.H. Kadlec, 2009). The characteristics that affect wetland nutrient removal capacity must be considered during design, construction, and management of these BMPs (Fisher and Acreman, 2004; Kadlec and Hey, 1994).

Floodplains offer a suite of characteristics that facilitate hydraulic and nutrient retention, such as wetland vegetation, low slope gradients, and potential for connectivity to the stream network (Bradley, 2002). These characteristics make floodplains desirable locations for created wetlands by enhancing connectivity to the stream and, subsequently, the filtration processes performed by wetland vegetation, microbes, and soils found in riparian areas (Tockner et al., 2010). Nutrient removal by constructed floodplain wetlands has been reported often in literature (Carleton et al., 2001; Moustafa et al., 1996; Noe and Hupp, 2007), making them practical options as BMPs for stormwater management in the proper hydrogeomorphic setting. Hydrology is the driver for many retentive processes that occur through the establishment and proliferation of wetlands. Groundwater and surface waters intersect at seep or slope wetlands that are commonly found in the floodplain of a stream channel (W.J. Mitsch, 2000). These intersections act as hotspots along the stream network for biogeochemical processing due to the presence of mesoand micro-scale energy gradients in redox potentials and continual nutrient loading from contributing flows, which are all affected by temporal cycles (Burt and Pinay, 2005; Tockner et al., 1999).

To gain an understanding of the establishment of a constructed wetland and the associated biogeochemical processes, hydrologic trends must be quantified on temporal and spatial scales. The dynamic pattern of saturation and drying caused by a fluctuating water table facilitates nutrient retentive processing performed by both anaerobic and aerobic microbes (Reddy, 2008). Understanding this hydrologic pattern may better inform wetland creation or restoration where goals are to re-establish hydrology that would create environments that optimize microbial conversion of non-point source (NPS) pollutants in stormwater runoff (Rucker and Schrautzer, 2010). To this end, it is beneficial for the wetland designer to quantify hydrologic inputs and losses and evaluate hydroperiod fluctuations to estimate an effective wetland volume for both anaerobic- and aerobically facilitated nutrient transformations.

Recent studies on the extent and scale of floodplain groundwater and surface water exchange show a need to expand from the scale of traditional approaches on surface water nutrient fate and transport that focus on in-channel processes to encompass the active floodplain (Woessner, 2000). Due to the complex nature of groundwater and the period of time needed to characterize water-table fluctuations, few wetland studies have completely characterized the hydrologic budget to include groundwater exchange, despite its potential role in nutrient fate and transport (Bradley and Gilvear, 2000; Raisin et al., 1999) and the complex interactions of adjacent topography that affect riparian hydrology (Claxton et al., 2003; Winter, 1999).

As part of a larger study to implement and monitor innovative stormwater BMPs, we constructed a floodplain wetland in 2007 near Winchester, VA, along Opequon Creek. The objectives of this study were to 1) determine if the CW meets the hydrologic criteria for jurisdictional wetlands (USACE, 1987), 2) investigate the influence of the groundwater component in the annual hydrologic budget for the CW during the time period of this study, and 3) identify spatial and temporal variability in groundwater hydraulic gradients. The significance of this study is that the data collected describe the hydrology of an innovative nutrient management BMP, and these results will better inform design of constructed floodplain wetlands in terms of hydrology and hydraulic storage in the floodplain and connectivity between floodplain groundwater and receiving stream. Furthermore, the period of study began approximately six months after wetland construction was completed and captured the critical establishment period of the CW.

III.2 Methodology

Study Area

Hedgebrook Farm constructed floodplain wetland lies along Opequon Creek, just south of the City of Winchester, VA. The CW encompasses 0.2 ha of reclaimed floodplain pasture provided by the private landowner at Hedgebrook Farm. The contributing basin to Opequon Creek at this location is approximately 30 km^2; predominately cattle pasture with increasing residential and commercial development. At stream baseflow, the CW is disconnected from the stream, receiving hydrologic inputs of precipitation and groundwater only.

Soils within the study location are mapped as predominantly Massanetta silty clay loam derived from limestone, with less than 2% slopes and clay subsoil. The average annual precipitation for the area is 88 cm, but during the two years of this study, the area received 106 and 85 cm (through October only) respectively (NOAA, 2010). A 9-year period of record for discharge of Opequon Creek is available at USGS Gage 01683450, located 2.4 km downstream from the CW. Peak discharge and mean annual daily discharge were 1.78 cms and 0.14 cms, respectively (USGS, 2009), for the period of study. Stream gage records show seasonal responses of the creek to precipitation and a discernable dry period surrounding the installation of the constructed wetland in May 2007. Persisting low water table elevations may still be a result of this 3-yr drought (Moorhead, 2003).

Constructed Wetland Design

The CW was built as a demonstration wetland with funding from the USEPA and administered by the National Fish and Wildlife Foundation for studies dealing with removal of nutrients from stormwater. The CW was designed using recommendations of the Virginia Department of Conservation and Recreation (VADCR) in constructed stormwater wetland guidance. This guidance was implemented through the integration of a variety of macrotopographic features with designed depths and surface areas within the floodplain area, including deep pools (1-1.3 m) for sedimentation, a low marsh (0.3 m) to increase flow path and contact time, and high marsh areas (0-0.1 m) for intermittent inundation (VADCR, 1999; VADCR, 2010). Established wetland vegetation in the floodplain was observed at an elevation that corresponded with the bankfull elevation in the stream channel as identified by field observations, channel cross-sectional geometry, and a flood frequency analysis of gage data that were area-weighted to the study site. The high marsh elevation was set to match this elevation (USDA NRCS, 2008). A berm was formed around the CW to enclose the area and to channel overland flows of runoff from the adjacent hillslope and upland areas around the wetland (Fig. 3.1).

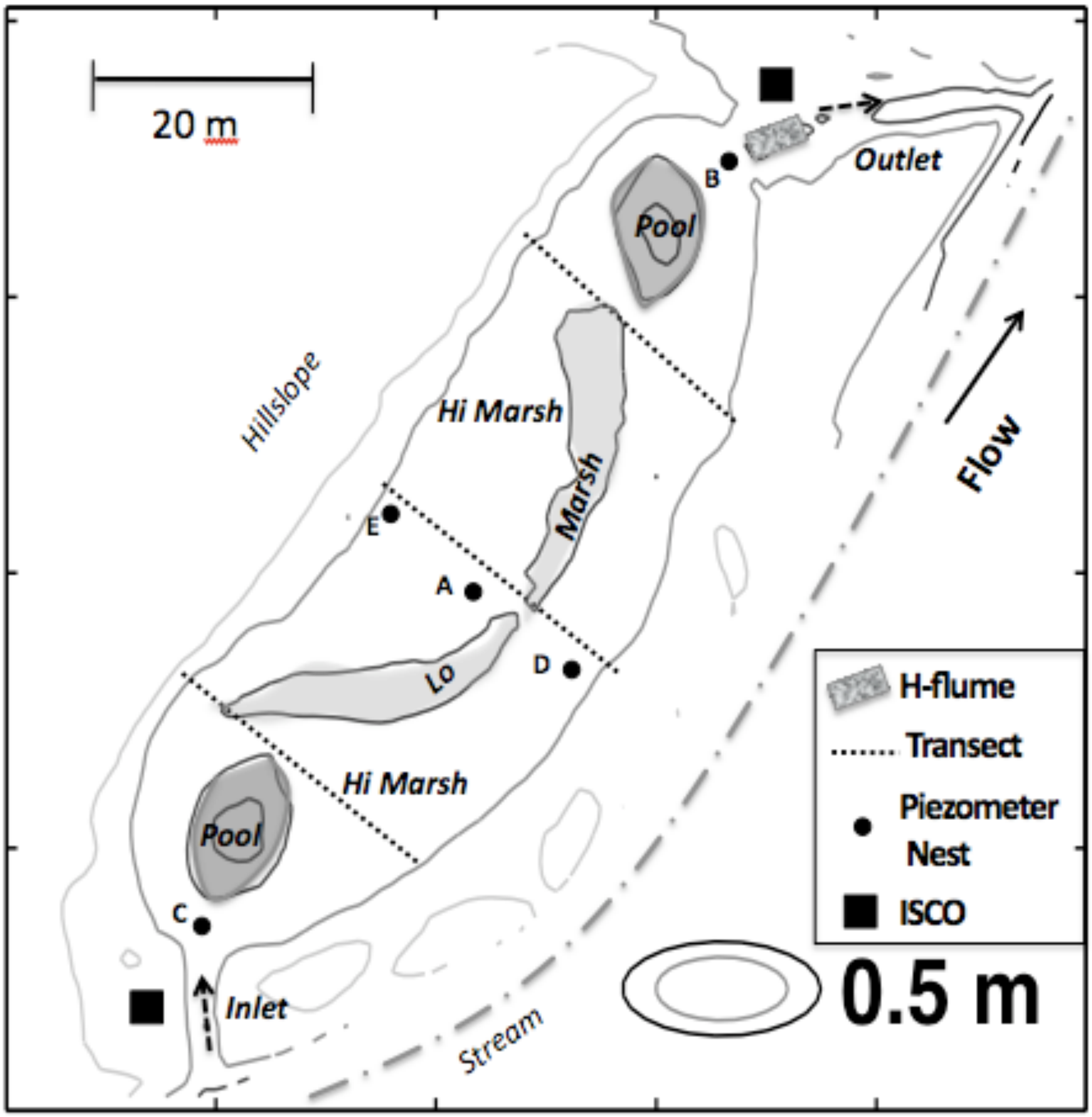

Figure III.1 Constructed wetland topography with piezometer locations and flow direction orientation.

Construction practices called for as little disturbance as possible during excavation, particularly in the high marsh area. After excavation, a final till was performed to decrease the effects of compaction on the soil surface and a mix of native wetland grasses and annual rye was immediately spread as the cover crop. Five months later, native wetland species were planted in delineated areas based on design water depth and anticipated inundation patterns and included *Scirpus validus* (bulrush), *Pontederia* *cordata* (pickerelweed), *Sagittaria latifolia* (arrowhead,) *Acorus calamus* (sweetflag), and a native wetland grass seed mix. These emergent species have rooting depths between 25 and 45 cm. The established wetland has a total surface area of approximately 1300 m^2, a wet-season baseflow volume of 100 m^3, and a volume of 250 m^3 when outflow through the H-flume is at capacity (0.3 m deep).

Field Measurements

Five nests of groundwater piezometers were installed within the CW in January 2008. Piezometers were constructed of 3.81-cm diameter solid PVC of various lengths (length required to reach datum plus a significant riser to extend through tall grasses), a 15-cm long slotted PVC section (0.025-cm slot thickness) that was coupled to the riser, and an inset PVC drive point at the end. Water-level measurements were recorded at each piezometer datum, which was the elevation of the 10-cm long slotted pipe portion below the riser (approximately 5 cm of the slots were closed due to glue from coupling). Piezometers were installed in the wetland by coring to the desired depth with an 8.9-cm hand auger, inserting the piezometer, backfilling with coarse sand to a depth above the slot, backfilling to within 15 cm from the surface with the cored soil, and then capping with bentonite to provide a water-tight seal around the PVC riser to eliminate preferential flow to the subsurface. Piezometers were then developed by the addition of a slug of water to flush out small particulates.

Each piezometer nest contained three piezometers; a deep piezometer that extends below a characteristic clay layer, a middle piezometer that is located in a thick clay layer, and a shallow piezometer that is above the clay layer (Fig. 3.2). Absolute pressure and temperature measurements within each piezometer were continuously monitored with water-level loggers (Onset Corp., Bourne, MA; accurate to 0.5 cm hydraulic head). An additional probe was incorporated in October 2008 and recorded atmospheric pressure in a mock piezometer casing in the wetland berm. Previous to this, atmospheric pressure was measured in dry piezometers and at a proximate weather station (6.4 km away) and the atmospheric pressure at the wetland site was determined through a relationship between the two during a period when there were no dry piezometers.

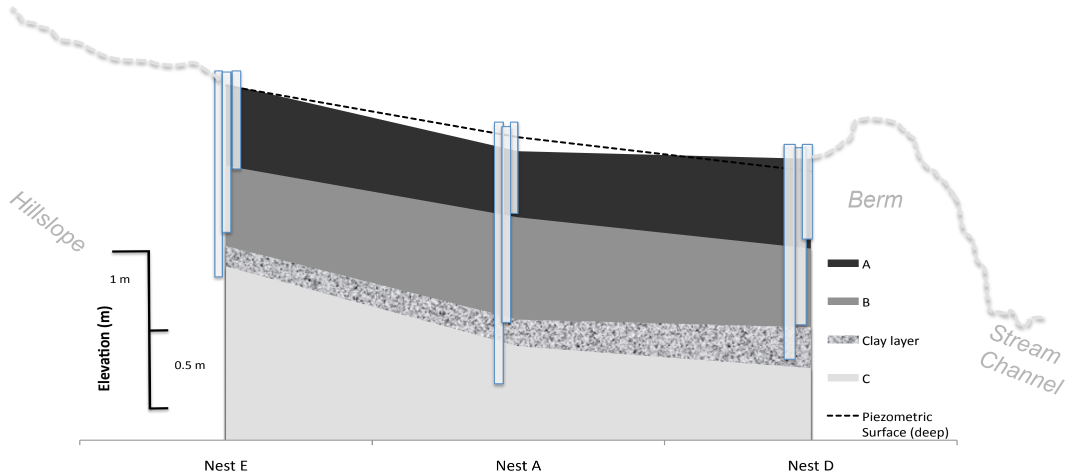

Figure III.2 Perpendicular transect of nested piezometers with associated soil profiles and water levels from May 2008.

Locations of piezometer nests were determined by selecting a central location in the high marsh and using triangulation over the wetland surface area, capturing perpendicular and parallel flows from hillslope across floodplain to stream, and along the hydraulic gradient down the floodplain. Specific depths of individual piezometer datum were selected by field observations during soil coring of change in soil profile layer as indicated by a change in color and field texture (Table 3.1). The objective was to capture the influence of this change in texture on groundwater movement.

Table III.1
Nested piezometer datum and soil layer information.

Nest	Piezometer	Depth (m)	Datum* (m)	Soil Layer Description
A	1 - deep	1.47	-1.47	Clay, dispersed sand and gravel
	2 - middle	1.04	-1.11	Heavy clay
	3 - shallow	0.44	-0.43	Silty clay loam
B	4 - deep	1.31	-1.55	Sandy clay
	5 - middle	0.89	-1.16	Heavy clay, dispersed sand
	6 - shallow	0.53	-0.78	Sandy clay loam
C	7 - deep	0.95	-1.01	Clay, some small gravel
	8 - middle	0.63	-0.66	Heavy clay
	9 - shallow	0.37	-0.47	Clay loam
D	10 - deep	1.28	-1.33	Clay, dispersed sand and gravel
	11 - middle	1.00	-1.10	Heavy clay

Nest	Piezometer	Depth (m)	Datum* (m)	Soil Layer Description
	12 - shallow	0.50	-0.59	Clayey sand loam
E	13 - deep	1.22	-0.80	Clay
	14 - middle	0.95	-0.53	Clay
	15 - shallow	0.54	-0.14	Clay loam

* Datum is the elevation of the point in the subsoil profile where water levels were read in each piezometer; the central Nest A – Peizometer 1 ground is at zero elevation.

Mapping

Wetland topography was captured using a Total Station (Topcon Corp., Tokyo, Japan) and described with 422 survey points. Piezometers were surveyed at the surface, and the datum were corrected to correspond to the topographic survey. This placed all piezometers in the same plane of reference. The error associated with surveying and determination of piezometer datum elevations was within 2 cm.

Hydraulic Conductivity

The Hvorslev method was used to determine the hydraulic conductivity of the soil layer at a depth of 1 m. Slug tests were performed with the addition of a slug of 1 L of water into piezometers at depths of approximately 1 m in Nests A, B, and D. Pressure measurements were logged every 20 s after slugs were introduced into the piezometer casings. Absolute pressures in piezometers that did not receive the slugs were also measured and recorded every minute. The slug response was established between the time of peak of the introduced slug and once the water level had equilibrated. There was no measured rainfall during the slug testing.

Water Table and Hydroperiod

The time series of pressure data collected in the piezometers was manipulated using MATLAB (Mathworks, Natick, MA). Hydraulic head in each piezometer was computed by subtracting the recorded atmospheric pressure from the measured absolute pressure and converting to head determined piezometeric surface in each piezometer. Water-level elevations were then determined by adding this value to respective datum elevations. Daily averaged pressure measurements were used in analysis of annual hydroperiod trends, while hourly data were used in analysis of event-specific responses. Data records were filtered to remove measurements of pressure when probes were removed from the risers for downloading and when measurements reflected less than 2 cm of head to account for dead space at the bottom imposed by the threading of the drive point.

Water table levels were analyzed in the middle Nest A to determine if the CW meets the jurisdictional delineation criteria for wetland hydrology, which is determined through the observation of the water table in the upper foot of soil for duration of the growing season (USACE, 1987). Hydroperiod seasons were delineated using statistical zonation, which compares the generalized distance within an analysis window that travels through the time series and uses the following equation:

$$D^2 = \frac{(\overline{a_1} - \overline{a_2})^2}{s_1^2 - s_2^2}$$ **Equation III.1**

where D^2 is the generalized distance that indicates shifts in trend, a is the middle of the data range within the analysis window, a_1 is data from $(a\text{-}h)\text{:}a$, a_2 is data from $a\text{:}(a+h)$, h is half of the selected analysis window, and s is the variance within the data (Davis, 2002). The Euclidean distance zonation was also employed as a second method to define hydrologic temporal zones using the following equation:

$$E = (\overline{a_1} - \overline{a_2})^2$$ **Equation III.2**

Hydrologic Budget

Hedgebrook CW operates as a demonstration and experiment wetland where inflows and outflows can be controlled and overland runoff does not reach the wetland by design. During the study period, there were no significant overbank flows. A deductive approach was utilized to estimate the magnitude of the groundwater component in the hydrologic budget. The water balance equation was simplified by assuming no groundwater exchange and calculated with the following:

$$S = P - ET$$ **Equation III.3**

where S is wetland storage (L), P is precipitation (L), and ET is evapotranspiration (L). Change in storage was calculated on a daily time step and cumulative storage was the resultant summation over the study period. Climatic data were accessed via the web from the National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center (NCDC). Precipitation and air temperature were collected at a Winchester, VA station (COOP ID 449181), located approximately 6.4 km from the study site. Solar radiation data were accessed via the web and collected at the most proximate NOAA weather station, which was in Charlottesville, VA (COOP ID 441593) and located approximately 128 km from the study site. Daily ET (mm/d) was calculated with the Priestley and Taylor (PT) method (1972) using the following equation:

$$ET_0 = \frac{\alpha}{\lambda}\left(\frac{s(R_n - G)}{s + \gamma}\right) \qquad \textbf{Equation III.4}$$

where α is the PT coefficient, λ is the latent heat of vaporization (MJ/kg), R_n is net radiation (MJ/m^2-d), G is soil heat flux(MJ/m^2-d), s is the slope of the saturation vapor pressure based on average daily temperatures (calculated as the mean of the maximum and minimum measured temperatures) (kPa/°C), and γ is the psychrometric constant (kPa/°C). The PT coefficient was selected as 1.26 for well-watered short grasses in humid regions (Lhomme, 1997). G was assumed to be negligible considering the daily time step of the analysis, assuming that the energy needed to heat the soil is the same energy lost to cool the soil in a day. In the absence of short- and long-wavelength radiation data, a method for determining R_n from commonly collected weather data was employed. Predicted R_n values were determined from an empirical relationship developed by Irmak et al. (2003) that used inputs of daily maximum and minimum temperatures, measured total solar radiation, and the inverse distance from the sun. The PT method was selected over other ET estimation methods due to the ease of its use with the meteorological data available for the region and the experiment support in literature for its use in well-watered short grass systems in humid regions (McAneney and Itier, 1996).

Storage was also calculated using a stage-volume relationship for the wetland. Surface storage volume was computed as the difference between a selected water-surface elevation and the existing wetland soil surface using the surface utility in AutoCAD Civil 3D (AutoDesk, San Rafael, CA). The resultant stage-volume relationship was described as log function (Fig. 3.3) and utilized in the second storage calculation. This relationship applies best when the water table is between 224.86 m and 225.31 m and may under estimate or over estimate storage at water tables lower or higher than this range, respectively.

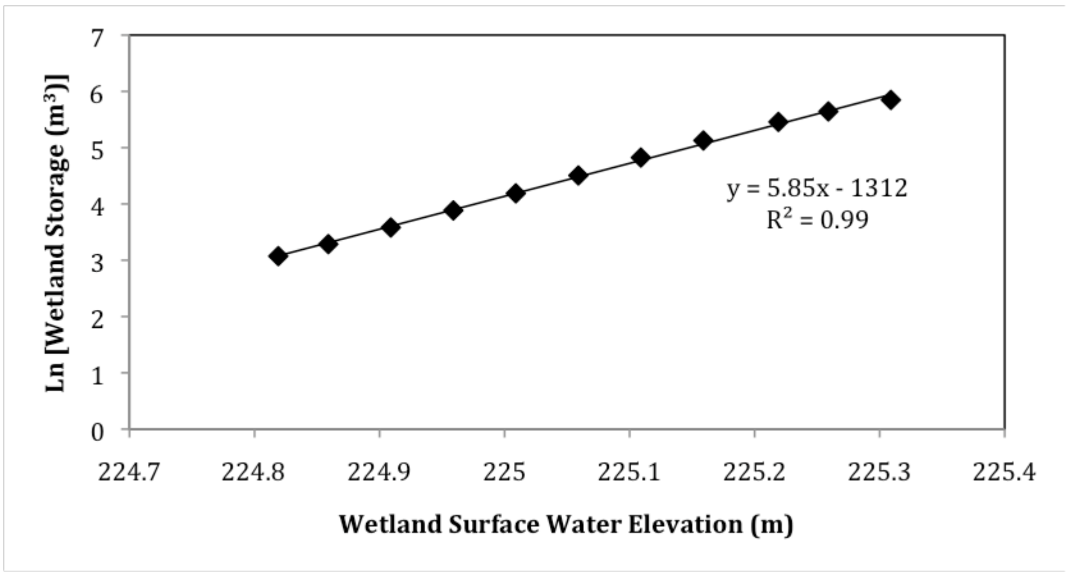

Figure III.3 Stage-volume relationship for the constructed wetland based on cut/fill analyses.

To investigate the adequacy of the tested water budget (with no groundwater exchange), the calculated P-ET budget was compared to the observed hydroperiod in the center of the wetland (which reflects groundwater exchange). The difference between the two time series provides an estimation of the role of groundwater exchange (discharge and recharge) within the hydrologic budget. The cumulative precipitation less the cumulative evapotranspiration time series was compared against the hydroperiod measured in the middle deep piezometer to assess this role.

Hydraulic Gradients

Lateral two-dimensional gradients were calculated between piezometer nests, comprising four adjacent triangular piezometric planes. These four planes are identified using a counter-clockwise convention between nests and labeled as planes EAB, ECA, DBA, and DAC. Groundwater gradients with flow components *i* and *j* in the planar *x* and *y* directions were spatially differentiated with the following relationship (Freeze, 1979):

$$v = \frac{\partial z}{\partial x} i + \frac{\partial z}{\partial y} j$$ **Equation III.5**

where ν is the hydraulic gradient, δz/δx (m/m) and δz/δy (m/m) are flow components in the *x* and *y* directions, respectively, corresponding with the field survey orientation and calculated with the following relationships (Abriola and Pinder, 1982):

$$\frac{\partial z}{\partial x} = \frac{(z_1 - z_2)(y_2 - y_3) - (z_2 - z_3)(y_1 - y_2)}{(x_1 - x_2)(y_2 - y_3) - (x_2 - x_3)(y_1 - y_2)}$$ **Equation III.6**

$$\frac{\partial z}{\partial y} = \frac{(z_1 - z_2)(x_2 - x_3) - (z_2 - z_3)(x_1 - x_2)}{(x_2 - x_3)(y_1 - y_2) - (x_1 - x_2)(y_2 - y_3)}$$ **Equation III.7**

where *x* (m) is the surveyed easting, *y* (m) is the surveyed northing, and *z* (m) is the observed water level elevation, each in the respective piezometer. Deep piezometers had the most complete record and, because of this, were used in all gradient calculations.

Flow magnitude and direction were calculated using Equation 8 and 9 (Mouser et al., 2005):

$$v_{mag} = \sqrt{\left(\frac{\partial z}{\partial x}\right)^2 + \left(\frac{\partial z}{\partial y}\right)^2}$$ **Equation III.8**

$$v_{dir} = \tan^{-1}\left(\frac{\frac{\partial z}{\partial x}}{\frac{\partial z}{\partial y}}\right)$$ **Equation III.9**

III.3 Results and Discussion

Hydraulic Conductivity

Saturated hydraulic conductivity varied within two orders of magnitude between nests when measured at a depth of just over a meter. All values were consistent with K_{sat} values for clay loam soils (Brady and Weil, 2002; Table 3.2). The nest at the wetland center (Nest A) exhibited slower K_{sat} than the streamside and outlet nests (Nests D and B). This supported field observations of sandier soils in the outlet area and water-level elevations were relatively larger losing hydraulic gradients. Finally, the faster K_{sat} measured at the streamside relative to the middle next may be due to increased coarse material that may be found in fluvial deposits in the natural berm of the creek.

Table III.2
Saturated hydraulic conductivity estimates from the data collected using a falling head slug test in piezometers.

Nest	K_{sat} (cm/s)	Depth (m)	Datum (m)	Location
A	3.91×10^{-6}	1.04	-1.11	Center
B	1.39×10^{-4}	0.89	-1.16	Outlet
D	5.57×10^{-5}	1.00	-1.10	Streamside

Hydroperiod

Water levels in shallow piezometers were used to delineate periods of connectivity between surface water storage and ground water in the constructed wetland. With the absence of a confining layer within the top portion of the wetland soil column, the piezometric water level in these shallow piezometers indicated the general water table level. The middle and deep piezometer datum are located at or below a change in soil texture (thick clay) and were used to evaluate vertical movement of groundwater. Nest A in the center of the wetland was selected to characterize the general hydroperiod of the constructed wetland.

Piezometric water levels indicated a connected free water surface and groundwater table. Trends in rise and fall in response to inputs from precipitation and surface flows occurred at the same time in all three nested piezometers at all five nests, indicating that the connected water table moved through the soil profile and across the floodplain without confinement. Piezometric water levels were consistently highest in the central Nest A, excluding Nest E, which lays at a higher elevation than the other four nests which are in the high marsh. This may be attributed to the confluence of the flowpath from the hillslope to the stream and the flowpath down the floodplain gradient that occurs in the wetland center at Nest A.

The constructed wetland meets jurisdictional hydrologic criteria defined by the USACOE. The criteria states that wetland hydrology may be established with observation of the water table within the top 0.3 m of the wetland surface for a duration of the growing season (USACOE, 1986). Water levels in the shallow piezometer in the wetland center recorded the water table at an elevation within the top portion for a significant duration of the area's growing season (Fig. 3.4). The growing season began on April 29th near Winchester, VA (UVA Climatology).

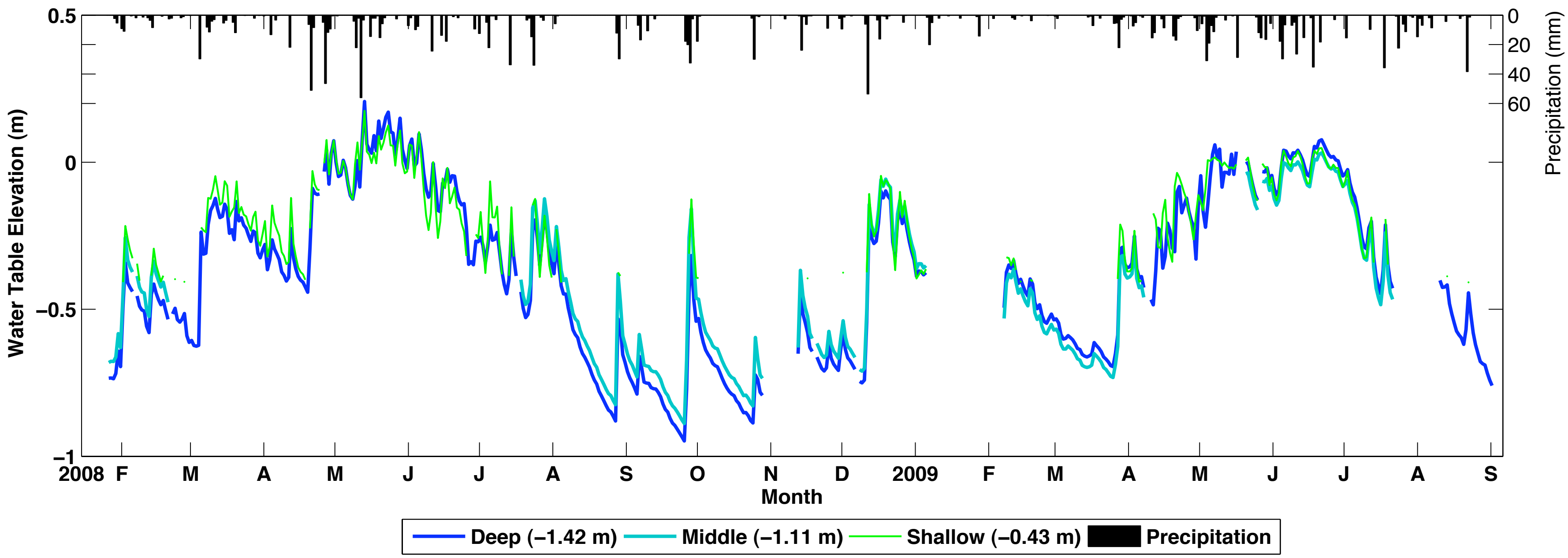

Figure III.4 Daily average water level elevation in central wetland piezometer Nest A. The ground elevation of the deep piezometer is the zero datum.

Groundwater tables also influence availability of dissolved nutrients to vegetation and the atmosphere above. Vegetation uptake of nutrients (Reddy et al., 1999) and oxygenation of subsoils through rooting (Neubauer et al., 2005a) have been shown to play critical roles in the fate and transport of nutrients in wetlands. The rooting depth of the planted native wetland vegetation in the CW is between 15-23 cm. During this study, the water table was measured to be continuously within the vegetation rooting depth in the months of March-July. This period of frequent precipitation and vegetation growth is a time of active conversion and sequestration of nutrients from stormwater and groundwater by wetland vegetation and microbes.

The CW water table response to temporal variations over an annual hydroperiod was described by delineating seasons using statistical zonation. The generalized distance method was used to delineate periods of changing variations in water table elevation created by continuous periods of either increase or decrease, while the Euclidean distance method was used to delineate changes in mean water table elevation. The confluence of these methods resulted in the delineation of two main seasons: 1) the wet season between March through July, when groundwater levels were highest, and 2) the dry season from mid July through December, when groundwater levels were lowest and event-driven patterns were prevalent (Fig. 3.5). These 'wet' and 'dry' seasons are referenced throughout the rest of the paper.

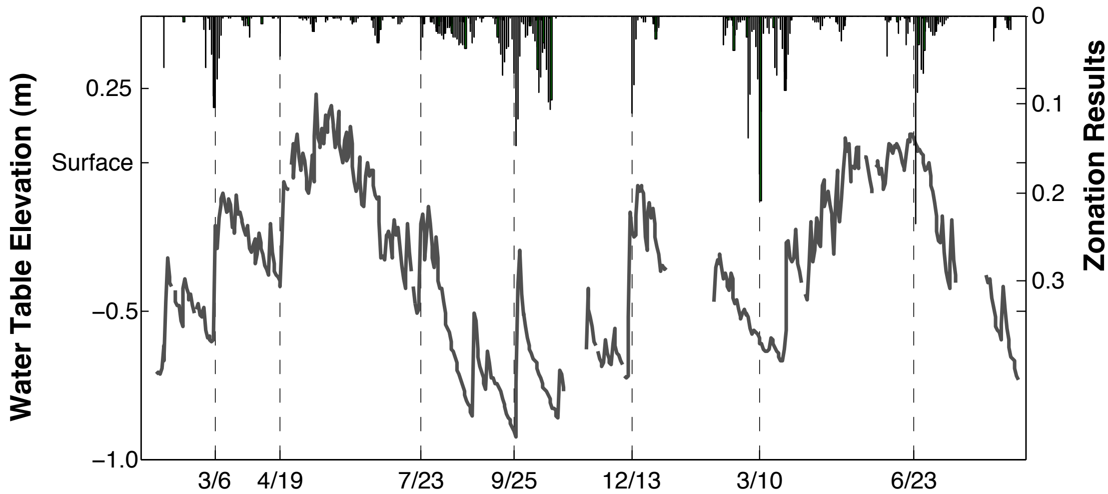

Figure III.5 Delineation of hydroperiod seasons resulting from statistical zonation analysis on Piezometer A1 time series water table data. Generalized distance (D^2) reported as Zonation Results on secondary axis.

Wetland Water Budget

Daily precipitation data were gathered from the NCDC. The period of record of available data was May 1982- Mar 2010. Annual precipitation trends were examined for the years of 1983-2009 to investigate the occurrence of trends present in the years of this study (2008-2009). The 27-year average annual precipitation was 976 mm of rainfall, where that of the study years 2008 and 2009 were 1085 mm and 1060 mm, respectively. However, during the winter months of Jan-Mar, the average rainfall was 207 mm, where that of the study years was only 158 mm and 110mm. Furthermore, during the spring/summer months of May-Aug, the average rainfall was 360 mm, where that of the study years was 413 mm and 512 mm. This comparison of the historical annual trends to those of the study years shows that the precipitation trends were not typical for the region (Fig. 3.6). This most likely impacted the annual hydroperiod.

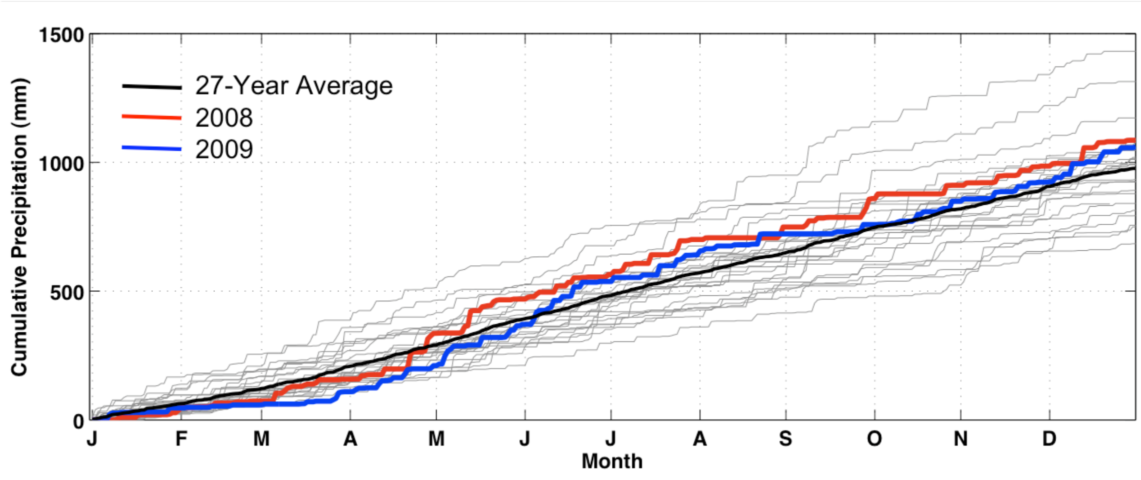

Figure 3.6 Annual precipitation trends of the years of study compared to the 27-year average cumulative precipitation trend.

The annual hydrologic budget was calculated using the tested storage budget, considering inputs of only precipitation and losses of only evapotranspiration. Groundwater exchange was not considered in the budget; however, comparisons to field measurements were used to evaluate the necessity of groundwater exchange in the budget calculation. Cumulative fluxes of precipitation and evapotranspiration exhibited seasonal trends that resulted in temporary net surplus or net loss in ΔS. Throughout the study time period, these fluxes were approximately equal in magnitude during an annual cycle (Fig. 3.7).

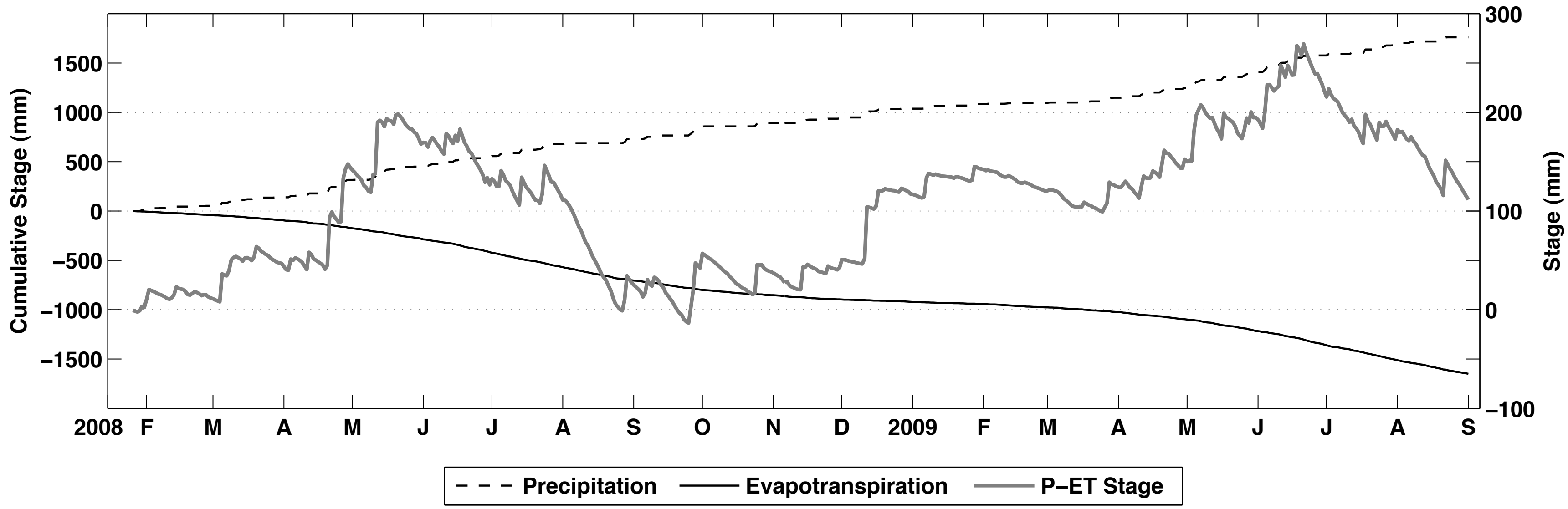

Figure III.7 Water budget components of precipitation and evapotranspiration and changes over time. P-ET Storage (ΔS) calculated as cumulative precipitation less evapotranspiration on a daily timestep.

In the months from February through July, the budget indicates a net surplus of ΔS, showing that the amount of precipitation is greater than the loss due to ET (Fig. 3.7). Low ΔS from August through November indicated evapotranspiration losses were and precipitation inputs were similar in magnitude. ΔS was at a minimum in late September. ΔS rebounded by the beginning of December, and a surplus occurred again through the beginning of the next year and until the end of the study period, where the tested budget estimated net ΔS surplus.

Field measurements determined that the water table was highly fluctuating, and ΔS was largely dependent on precipitation. These observations also show increased storage from May through June that is a probable result of groundwater inflow from deep and local recharge following precipitation events. Low water table levels persisted in the months of August through December.

The P-ET budget indicated similar general seasonal trends in storage as observed in the field measurements in the Nest A piezometers (Fig. 3.8). However, the P-ET budget did not match the hydroperiod in either of the two growing seasons covered in the time period of this study. The water table fluctuated much more than the P-ET budget would have estimated for the site. While the P-ET budget calculated an annual high of 0.260 m storage above a datum and an annual low of -0.013 m below a datum, the water table was measured to be at a high of 0.206 m and low of -0.984 m. The fluctuation range produced by the P-ET budget was 0.282 m, while the range of the water table observations was 1.19 m. This has large implications for vegetation establishment and water quality treatment. A hydroperiod fluctuation of a meter would produce conditions that are either too dry or too wet for vegetation if they were selected based on a P-ET budget at this site. Furthermore, the water treatment potential of the site decreases when the water table is not within the root zone of the vegetation (Tanner, 1996; Yang et al., 2001). These findings indicate the importance of understanding the local groundwater hydrology of a site if water treatment or vegetation grow-out is expected.

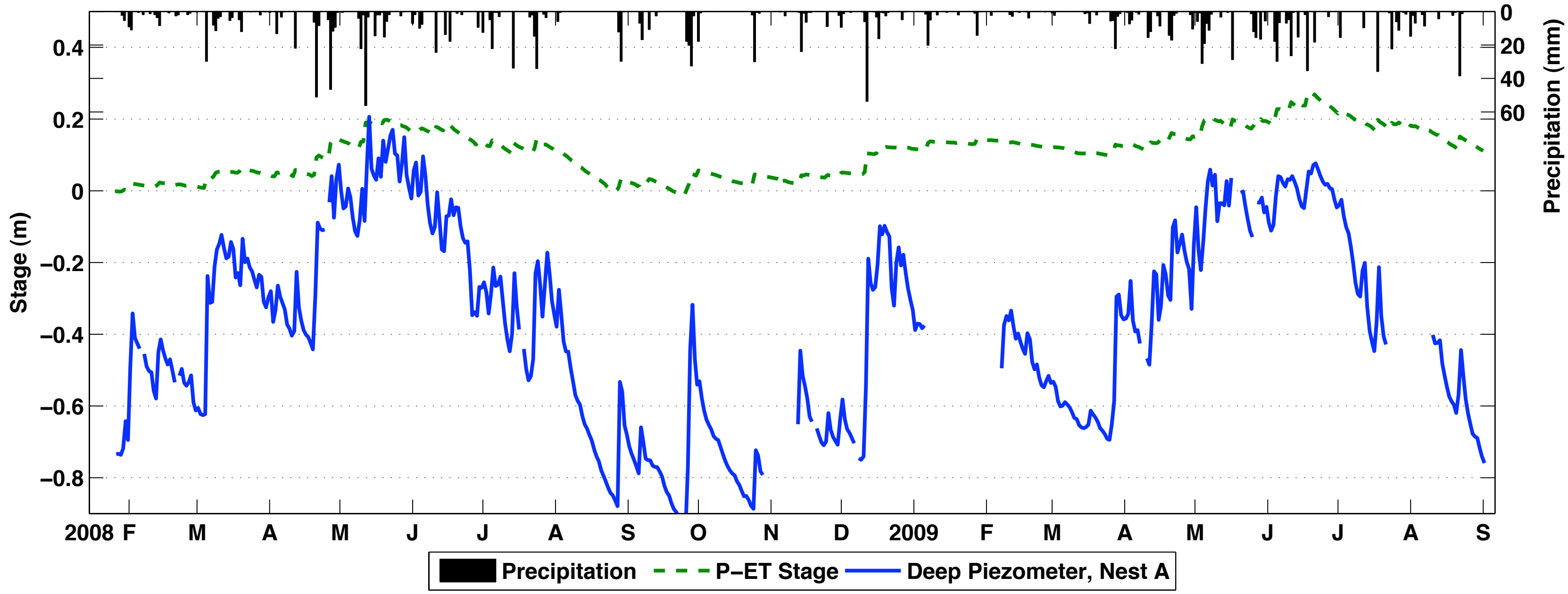

Figure III.8 Water budget (P-ET) as compared to observed water table fluctuations in the deep piezometer in Nest A. P-ET is assumed to start at a relative zero at the beginning of the study period.

The tested budget was compared to the field measurements on a seasonal basis, and it was found to under estimate wetland storage during the spring and into summer, while it over estimated storage during the rebound of storage during the winter that followed the dry season. These observations show that during the growing season, there may be too much storage depth on site for wetland plants if the species is sensitive to standing water and inundation patterns and if these species were selected based on the tested budget. Furthermore, losses to groundwater seepage following the dry season are not accounted for in the tested budget. This may affect the storage predictions during the winter months and lead to an incorrect estimate of the water table surface during the early spring. If groundwater exchange was truly negligible in the hydrologic balance, the tested budget predictions of storage should be mirrored in the observed field measurements in the unconfined piezometers during all seasons of the hydroperiod. There was no documented surface inflow, and surface outflow only occurred briefly after precipitation events. Given these observations, we determined that groundwater exchange within this constructed wetland system significantly affected the water balance and hydroperiod, and incorrectly characterizing groundwater exchange would affect predictions of water table elevation and storage during much of the year.

Rounded peaks and broad bases characterized storage response in the second year, while in the first year of observation, steep spikes characterized responses with large variability. This could be attributed to filling of previously empty pore spaces during the first wet season after construction in the first year and continued inundation through the second year. Ultimately, changes between years one and two resulted in a damped response of the groundwater surface to changes in inputs.

Hydraulic Gradients

Hydrological patterns observed in the CW indicate that the wetland functions as a seep or riparian wetland; it is located at the base of a sloped area where the groundwater surface intersects the land surface and it discharges water downstream as surface flow or groundwater (Mitsch and Gosslink, 2002). The CW is generally recharging groundwater in the dry season, when losing vertical gradients were observed in all piezometer nests, and discharging groundwater in the wet season, when gaining vertical gradients were observed most notably in the central Nest A. Following precipitation events, extreme downward gradient was documented at the streamside Nests D and C. These gradients may be a result of increased sand in the soil profile moving from the hillslope to the stream. Periods of highest piezometric water level in the middle horizon piezometer showed that interflow may have been local recharge as infiltrated water hit a layer of lower hydraulic conductivity and moved down slope via gravity. This was observed particularly during the summer months in central Nest A.

Water levels were consistently highest in the hillslope Nest E and showed the greatest lateral hydraulic gradient to be along the transect perpendicular to the stream, from hillslope to the channel (Fig. 3.9). In the lateral transect, piezometric water levels showed a very modest gradient from the wetland inlet to the outlet: however, these two transects intersect in the wetland center at Nest A, where groundwater potentials are greatest amongst the high marsh nests. Generally, flow is moving from hillslope to stream and laterally along the floodplain from the inlet area of the wetland toward the outlet (Fig. 3.9). Water levels at the outlet Nest B were lower than the other nests in the high marsh, in both relative elevation and depth below surface, suggesting a slight change in substrate size between the middle transect and the outlet. Field observations indicated surface water elevations in the area of Nest B were consistently lower than that of the forebay of the wetland. Soils were described as sandier in the lower part of the floodplain than those in the upper and middle areas, and hydraulic conductivity was higher at Nest B (Table 3.2).

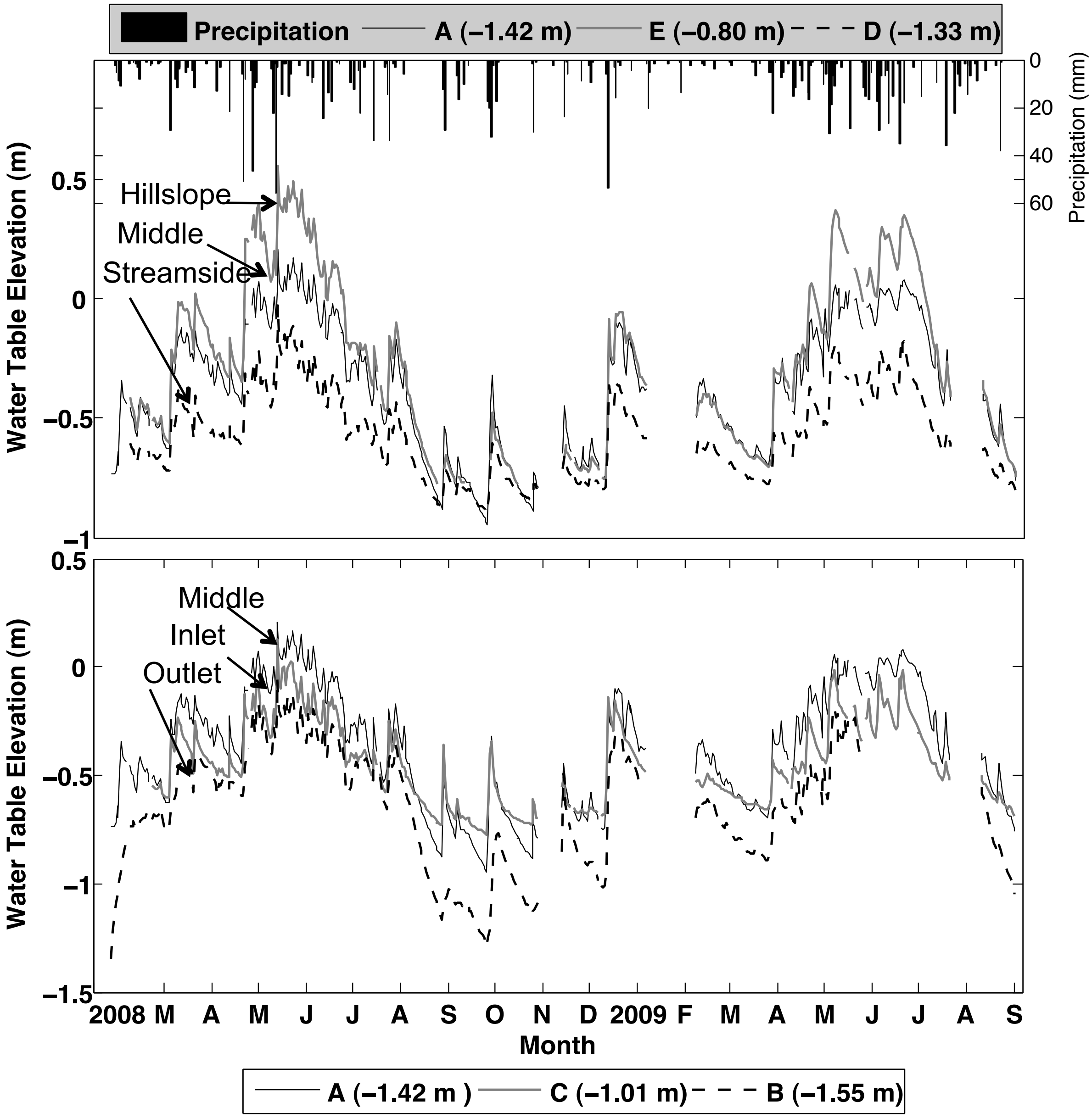

Figure III.9 Water levels in deep piezometers within five nests and compared within the perpendicular transect (top) and parallel transect (bottom) to the stream channel.

In the central Nest A, recharge hydraulic gradients were measured from the beginning of the study period (Jan 2008) until May 2008, when a discharge gradient began to build through the end of June (Fig. 3.10). Recharge gradients were again measured from July through the following February. Winter months are typically periods of discharge, as there are minimal ET losses during low temperatures and vegetation dormancy. However, in the uncharacteristically dry winter, recharge gradients persisted later into the winter months than expected. Discharge gradients were measured again in February and persisted through the early spring. Beginning in April, dynamically changing gradients between the deep and shallow piezometer were measured, possibly reflecting an increase in plant transpiration, rainfall, and daily average temperatures.

Water levels measured in the piezometers responded quickly to rainfall events. A response in water table elevation and hydraulic gradient was observed typically within the same day as the measured rainfall. The response was also quick to tailoff, producing the spikey hydraulic gradients shown in Fig. 3.10. Particualarly in spring 2009, the vertical hydraulic gradient measured between the deep and shallow piezometer showed downwelling occuring right after measured precipitation followed by a quick shift to upwelling within 2-3 days. This fast response time to precipitation may be attributed to the hydrogeologic framework of the area, which is characterized by karst features such as sinkholes, springs, poorly developed surface drainage over carbonate bedrock (Orndorff, 2002). The watershed is also studied by the USGS in a national karst project.

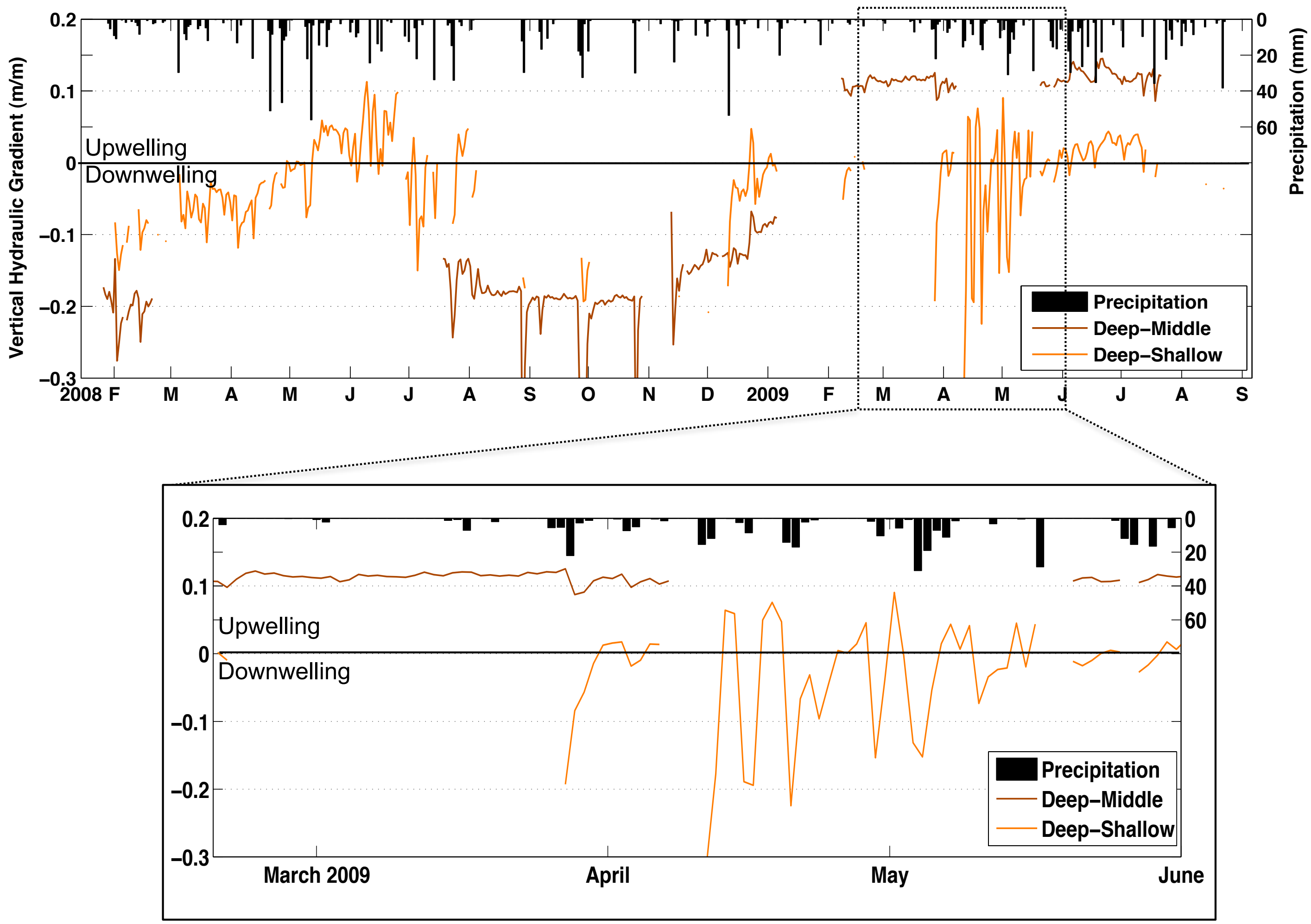

Fig. 3.10 Vertical hydraulic gradients measured in Nest A between deep, middle, and shallow piezometers. Portion of hydroperiod blown up to highlight fast response time in hydraulic gradient to precipitation (average daily gradients, total daily precipitation).

The magnitude of vertical hydraulic gradients (ranging from -0.3 to 0.1 m/m) was significantly greater than that of lateral hydraulic gradients (ranging from 0.0 to 0.02 m/m), and most often, by an order of magnitude. This suggests that high infiltration rates at the surface may dampens the effect of lateral gradients caused by topography. The dominant flowpath of water was vertically through the wetland substrate (Fig. 3.11). The greatest vertical gradient was measured closest to the stream, where great downward movement of water through the natural berm along the creek to the channel was expected. Combining the effects of highly fluctuating vertical gradients with shifts in lateral gradients indicates that the hydrology of this small CW with a low-sloping surface is very complex and highly variable on both spatial and temporal scales.

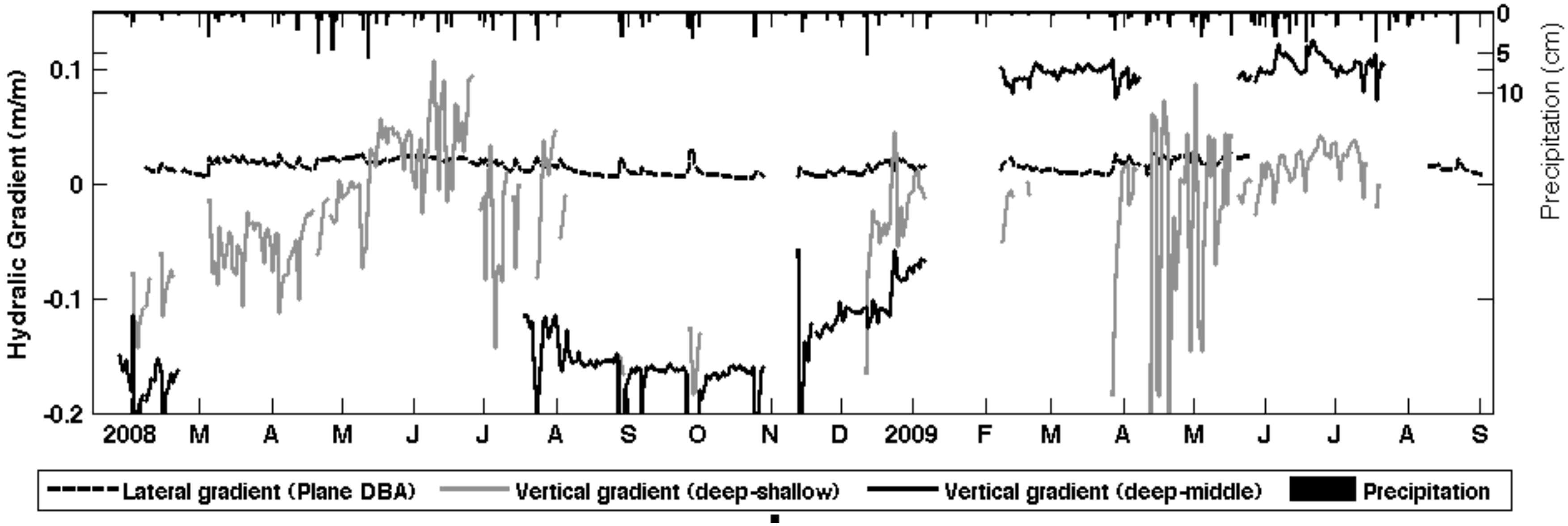

Figure III.11 Vertical hydraulic gradient between the deep and shallow piezometer in Nest A compared to the lateral hydraulic gradient from the wetland toward the stream channel (Plane DBA).

Triangulation was used to create piezometric planes between nests, resulting in four triangles that each shared two faces and mid-point of Nest A. In response to precipitation in the dry season, gradients near the hillslope in planes EAB and ECA had longer and more gradual rise to peaks that were generally higher than those near the stream channel in planes DAC and DBA, where responses were sharper and more immediate (Fig. 3.12). The hillslope planes also fall off sharply after peak, while the planes near the channel taper off gradually. These trends could be attributed to interaction of water draining from the hillslope and the stormflow in the stream channel. Infiltration and runoff affects the planes near the hillslope and occur at a gradual pace before the peak of a rain event. Rise in stream stage affects the planes close to the channel and occurs quickly to the peak of the event and then slowly recedes after the peak as storage drains back into the creek after being pushed into the floodplain substrates through the banks of the channel. In summary, the planes closest to the hillslope were influenced more by the hillslope-floodplain water-table fluctuations, while the planes closest to the stream were impacted by the changes in stream stage. These two effects combine in the floodplain to form wetland hydrology in the CW.

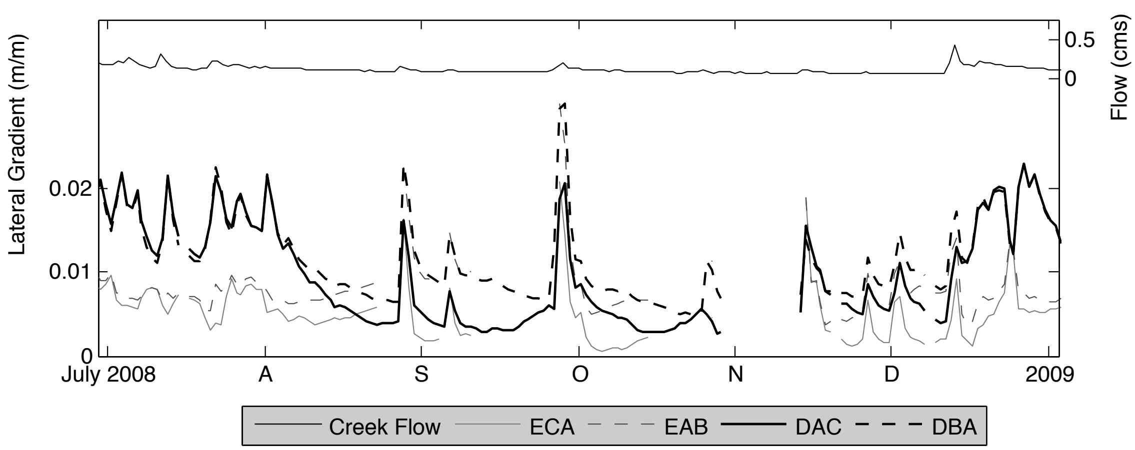

Figure III.12 Lateral hydraulic gradient calculated from water-level records in deep piezometers during the dry season and Opequon Creek flow record (USGS 01614830).

Generally, lateral gradients from hillslope to stream dictate the lateral water movement through the floodplain. However, in the dry season, flow directions become dynamic and oscillate frequently between flows to the stream channel and away from the channel. This may be a product of the low water table in the floodplain and the fluctuating stage of the surface water in the stream channel. Groundwater gradients are influenced by the rise in stage of surface water in the stream to create a ridge along the stream that would force water into the floodplain.

There is much evidence in the collected data indicating the wetland hydrology is driven by precipitation. First, the annual hydroperiod follows closely to the annual precipitation trends. The annual precipitation trends in these two years of study were not consistent with 27-year average annual trends – with little rainfall measured in the winter and above average in the summer. Correspondingly, the hydroperiod showed an annual minimums beginning in the fall and persisting through the winter, and annual maximums in the summer. This is opposite of the typical riparian wetland hydroperiod, where a rising water table through the winter months is expected due to minimal evapotranspiration, and a falling water table in the summer is expected when both temperatures and plant transpirations are at their annual peaks. Second, the reaction rate to precipitation is very high. Often within a day of an event, the water table was measured to rise in response. Finally, the large vertical gradients from the surface of the soil profile to the deep piezometer after measured rainfall suggests that the movement of water through the wetland is predominately that of fallen rain moving vertically through the soil column.

III.4 Conclusions

Within a year after construction, the CW was observed to meet the hydrologic criteria set forth by the USACE to delineate jurisdictional wetlands. The wetland water surface was fully connected to groundwater and responded to precipitation events with little evidence of major confining layers prevalent in the soil subsurface. The overall hydrology, storage, and movement of water in the wetland were driven by precipitation. The influence of rainfall events was evident in hydroperiod trends as well as hydraulic gradients throughout the wetland.

The hydrologic budget was predicted using common design practices that exclude the influence of groundwater exchange between the wetland storage and groundwater surface. Assumptions made for application of the PT method for ET would introduce error that would be then incorporated into the tested P-ET budget. The selection of a static alpha that was not empirically defined may lead to over-prediction of ET losses (Soucha et al., 1996). However, PT ET has been shown to adequately represent losses in saturated, short-canopy, humid systems (Drexler et al., 2004; Sumner and Jacobs, 2005).

When compared to the observed hydroperiod, it was apparent that this P-ET budget resulted in the underestimation of wetland storage and water table fluctuation, which are critical factors of wetland vegetation establishment and water treatment. The incorporated ET error is not expected to account for the discrepancies between the hydroperiod fluctuations (up to 0.6 m). All findings point to the importance of correctly characterizing local groundwater hydrology to understand the role of hydrology in the establishment and proliferation of wetland form and potential ecological function.

Patterns in hydraulic gradients indicate a significant impact of the adjacent hillslope that dominated the lateral flow of water during the wet season (March-July) and of the fluctuating adjacent stream stage that dominated in the dry season (August-December). Piezometer data also indicated great temporal variability in vertical gradient, which followed precipitation trends. These findings on vertical gradient variability and seasonal trends were consistent with other water table dynamics studies in Appalachian floodplains (Cole and Brooks, 2000; Moorhead, 2001).

Wetland inundation during the spring season as well as event-driven inundation periods were consistent with characterizations of floodplain hydrology in the Ridge and Valley of Pennsylvania made by Cole and Brooks (2000). Increasing clay soils with depth appeared to have an influence on the flow of water from the adjacent hillslope. These findings were also consistent with those of Moorhead (2001) in a Southern Appalachian floodplain. With a median depth to water measurement of -37 cm (maximum = 21 cm, minimum = -98 cm), this site would fall into the severely disturbed mainstem floodplain class as described by Cole and Brooks (2000). Given the atypical nature of the annual precipitation trends during the years of study, the depth to water measurements may have been effected by uncharacteristic periods of little precipitation during the winter months when discharge would be expected. With this consideration, more years of water table data are needed to clearly classify this floodplain wetland site within a disturbance classification.

Acknowledgements

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IV. Influence of Macrotopography on Flow Hydraulics in a Small Constructed Floodplain Wetland

IV.1 Introduction

Wetlands provide ecological services that are crucial to the sustainability of surface water resources. Studies have shown that natural and constructed floodplain wetlands have the capacity to attenuate non-point source pollutants from stormflow on the time scale of natural storm events (Casey and Klaine, 2001; Fisher and Acreman, 2004; Noe and Hupp, 2007; Schulz and Peall, 2001). Design recommendations for constructed stormwater wetlands call for the integration of different macrotopographic features into the design to control settling of suspended solids in specific areas, introduce flow complexity to slow moving water, and provide varying hydrologic patterns for a heterogeneous standing crop of emergent vegetation (ART, 1997; Schueler, 1992; VADCR, 1999; VADCR, 2010). These macrotopographic features along with wetland vegetation create spatial variability in hydraulic parameters that govern flow through wetlands.

Hydraulic transport processes in constructed wetlands influence the residence time of flow through these systems, which affects the fate and transport of solutes (Keefe et al., 2004). Modeling of constructed wetlands as non-ideal flow through reactors has gone beyond the limitation of the assumptions necessary to describe wetlands as either plug-flow reactors (PFRs) or fully-mixed reactors (or continuously stirred tank reactors, CSTRs) to incorporate axial dispersion and mixing (Kadlec, 1994). Wetland flow modeling has been used to check for short-circuiting (Crohn et al., 2005; Martinez and Wise, 2003) and determine pollutant removal rate coefficients that are impacted by the effective volume ratio of a wetland (Persson and Wittgren, 2003).

Hydraulic parameters are coupled with information about loading rates in inflow concentrations of pollutants to inform design of treatment wetlands (Kadlec, 2000). The challenge of hydraulic parameterization of a constructed wetland for performance prediction lies in capturing the spatial variability mechanisms that drive pollutant removal. Tracer injection experiments can help to quantify a variety of these hydraulic processes. Important factors to consider in performing an effective tracer experiment include 1) adequate flushing of previous tracer; 2) use of inert tracer; 3) adequate sampling frequency; and 4) accurate knowledge of wetland water volume (Werner and Kadlec, 2000). The objectives of this study were: 1) characterize hydraulic parameters of a constructed wetland using conservative tracer injection during controlled flooding events; 2) evaluate the capacity of wetland macrotopographic features and vegetation to introduce flow complexity and mixing; and 3) quantify the role of wetland macrotopographic features in the removal of suspended solids from overbank flows.

IV.2 Methods

Study site

The study was conducted on a constructed floodplain wetland that was implemented in spring of 2007 along Opequon Creek in the Ridge and Valley Physiographic Province of Northern Virginia. The site was designed as a demonstration best management practice (BMP) for management of flow and nutrients from overbank stormwater as part of an EPA Targeted Watersheds Grant for the Chesapeake Bay. The design was developed through the mapping of local topographic wetland characteristics and results of a flow frequency analysis for Opequon Creek (Fig. 4.1). A series of macrotopographic features of high and low marshes and settling pools were incorporated into the design area of about 0.2 ha (Table 4.1; VADCR, 1998). Native wetland emergent vegetation was established in the wetland site for two years previous to this study. The effective surface area of the wetland is approximately 1300 m^2, which corresponds to a storage volume of approximately 250 m^3. A stage-volume relationship was developed using the cut/fill tool in AutoCAD Civil 3D (Autodesk, Inc., San Rafael, CA) and used in calculations in which wetland surface storage volumes were needed.

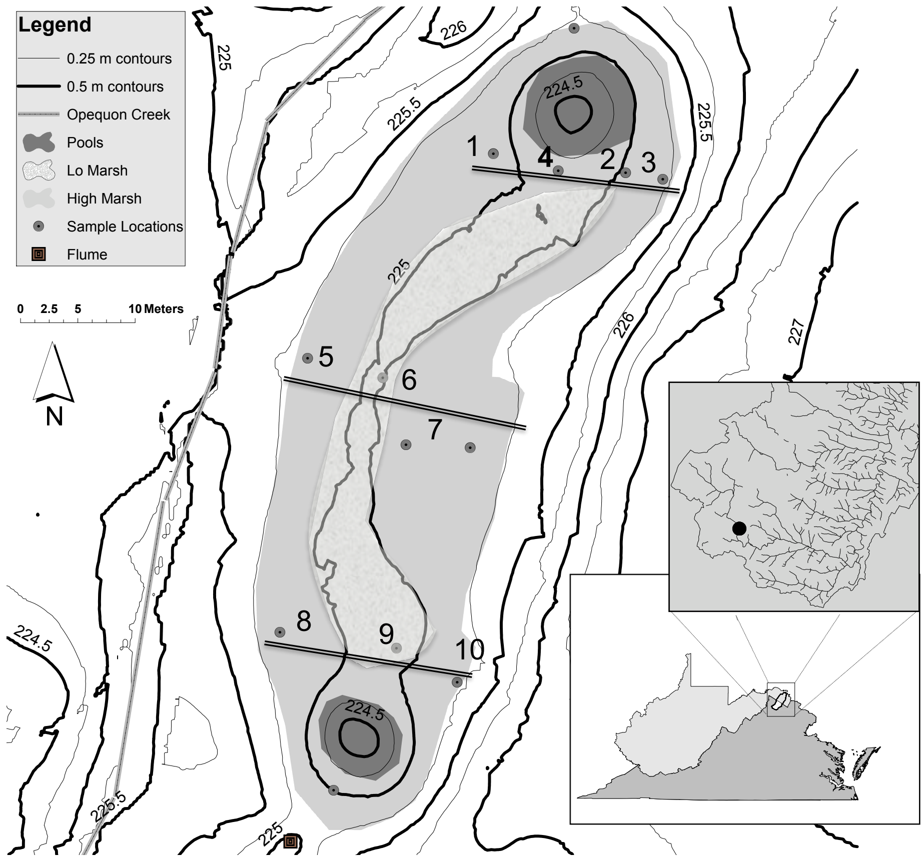

Figure IV.1 Hedgebrook Farm constructed floodplain wetland topography and tracer experiment sample locations.

Table IV.1
Wetland macrotopographic feature dimensions.

Wetland Feature	Depth Range (m)	Surface Area (m^2)
Forebay	0.7	69
Exit Pool	1.0	40
Low Marsh	0.2-0.3	280
High Marsh	0-0.2	910

The wetland was designed to provide the ability to monitor surface inflow and outflow as well as groundwater table. The wetland is surrounded by a berm that encompasses the wetland and creates the ability to conduct pumping experiments. The wetland level is controlled by a 0.3-m H-flume, and the inlet and outlet are instrumented with flow samplers. Nested piezometers are deployed at five locations throughout the high marsh and continuously monitor water levels at three different soil layers in the wetland soil profile. No rainfall fell during the pumping experiments.

Controlled Flood Events

Controlled flood events were conducted in two seasons at the constructed floodplain wetland site, one event in fall (November 2008) and a second event in spring (May 2009). The main objectives of the flood events were to collect hydraulic and nutrient attenuation data needed to gain an understanding of internal processes of the wetland. Stream water from the adjacent second-order stream was pumped from the stream channel with a trash pump. Potassium bromide (KBr) was used as a conservative tracer during a step response injection experiment and an impulse input experiment. KBr was selected because bromide is an inert tracer that does not react with its environment at ambient temperatures (Standard Methods 4500, APHA 2000) and has a large solubility factor, which makes it easy to create a well-mixed injectate solution. In step input experiments, KBr was pumped into the trash pump outfall using two calibrated variable speed injection pumps (Fluid Metering, Inc., Syosset, NY) with a pump rate of 88 mL/min to obtain a target bromide concentration of 10 mg Br/L. During the impulse experiment, a slug of KBr solution was introduced at the pump outfall all at once.

An inlet channel of 10 m acted as a mixing zone for the injectate solution and the pumped stream water. After traveling through this mixing zone, wetland inflow entered the high marsh of the wetland. The preferential flow path through the wetland based on elevation gradients pushed flow from the mixing inlet channel, through a small distance of high marsh area, then into the forebay. From the forebay, preferential flow follows a sinuous curve through the low marsh until flow is normal to a second patch of high marsh area. After jumping over the high marsh, preferential flow travels through a second and final pool before gliding over a second small patch of high marsh and then out of the H-flume control structure at the wetland outlet. These pools and marshes are the macrotopographic features that were designed to encourage mixing and flow complexity.

The wetland inlet and outlet were instrumented with automated sampling units called 6712 Portable Samplers (ISCOS, Teledyne ISCO, Inc., Lincoln, NE). The inlet unit comprised of a 750 Area Velocity Flow Module and automated sampling system, while the outlet unit utilized a 730 Bubbler Flow Module to capture stage in the 0.3-m H flume and an automated sampling system. Inflow was quantified using continuity, while a flume-rating curve quantified outflow based on stage measurements. Three sampling transects were installed normal to preferential flow and spanned the low and high marsh areas of the wetland (labeled A, B, and C). Ten sampling locations were established along these three transects, four along A and three along B and C.

Inflow was pumped for a total of approximately eight hours during the step input experiments, and for approximately seven hours during the impulse experiment. During the impulse experiment in May, inflow was pumped through the wetland for two hours to flush out the KBr from the previous day's step response experiment. The impulse injection was then performed and tracer response was monitored in the following 4 hours of pumping. For all experiments, tracer injection began once a "steady state" of flow was established, which was determined to be when the stage of the wetland was constant as measured by stage gages located throughout the wetland. Samples were collected at the inlet and outlet through the automated samplers that composited samples over 30-minute periods to collect 1 L of sample in acid-cleaned polyethylene bottles in the fall event. The ISCO units malfunctioned at the beginning of the spring event, making them unreliable for the inlet-outlet sampling. Instead, grab samples in the inlet and outlet were used in place of ISCO composite samples. Event volumes were calculated by integrating the average discharge measured over the time of each collected sample. Rapid sampling occurred at the ten sampling locations located along transects A, B, and C. Samples were collected through manual grab in 20-mL scintillation vials at intervals ranging from 5-15 min depending on distance and time from tracer injection. Total step input tracer time was 135 min in the fall event and 200 min in the spring event.

Starting water table elevations were measured the day before the event in nested piezometers spaced throughout the high marsh of the wetland. The water table at the center of the wetland was measured to be 80 cm below the wetland surface before the fall event and 5 cm before the spring event. The inundated area was delineated during each flood event with flags, surveyed using a Total station, and aligned to the previously mapped topography. This characterized the elevation of the high water line and was used to determine the slope of the wetland water surface. Storage in the fall event was 215 m^3 and 250 m^3 in spring.

Laboratory Methods

Samples were filtered in the lab using a 0.2 μm syringe-tip filter and a polyethylene syringe that was rinsed three times with ultra pure water (Millipore, Billerica, MA) between samples. Bromide concentrations in all samples were determined using a Dionex 3000 ion chromatograph (IC) consisting of an autosampler, injection pump, conductivity detector, and an IonPac AS18 4x250 mm analytical anion column. The IC unit was calibrated proximate to each determination using dilutions from purchased standards (RICCA Chemical Company) and a linear correlation from a five-point calibration curve (Dionex, Sunnyvale, CA). The practical quantification limit was 0.5 mg Br/L.

TSS was determined using filtration of a known volume of grab sample through glass fiber filters (Pall Life Sciences, Inc., Port Washington, NY)(Standard Method 2540; APHA, 2000). Filtrate volume of each sample was determined upon filtration as the volume passed through the filter in a reasonable amount of time and to collect a minimum of 2 g solids.

Tracer Response Analysis

Impulse experiment breakthrough curves were used to determine wetland mean residence time and dispersion parameters (Kadlec, 2009a). The nominal (theoretical) residence time during these events was calculated using the following:

$$\tau_n = \frac{V}{Q}$$ **Equation IV.1**

where τ_n is the nominal residence time (T), V is the wetland volume (L^3), and Q is the hydrologic load or flow rate (L^3/t).The cumulative residence time distribution (RTD) function *f*(t) was determined for an impulse input of tracer into a steady-state flow system, forming the cumulative residence time distribution *f*(t)(Kadlec, 2009a). Mean residence time was determined through the integration of the cumulative residence time distribution on a mass basis:

$$\tau = \frac{1}{M_i}\int_0^\infty tQCdt$$ **Equation IV.2**

where τ is mean residence time (T), Q is outflow (L^3/T), C is the tracer concentration (M/L^3), M_i is the total mass of added tracer, and t is time. The ratio of the mean residence time to the nominal residence time determined the volumetric efficiency:

$$e_V = \frac{\tau}{\tau_n}$$ **Equation IV.3**

Another parameter of interest was the spread of the tracer response curve about the mean residence time, and this variance is determined with the following:

$$\sigma^2 = \int_0^\infty (t - \tau^2) f(t) dt$$ **Equation IV.4**

From this variance, dispersion characteristics were determined. A dimensionless variance of the tracer was determined through dividing the squared variance by the squared retention time (τ). The inverse of this dimensionless variance gives the number of tanks (N, or shape parameter) needed to describe the tracer pulse in a tanks-in-series (TIS) model. This variance is also used to determine the wetland dispersion number:

$$\sigma_\theta^2 = 2\delta - 2\delta^2(1 - e^{-1/\delta})$$ **Equation IV.5**

where δ is the wetland dispersion number (dimensionless). This model can be used to quantify the axial dispersion from plug-flow systems. In the presence of large amounts of mixing, this model may not adequately describe dispersion (Levenspiel, 1972); however, this model was used to quantify dispersion in such a manner as to compare to other FWS wetland studies.

Step input experiment breakthrough curves were used to describe spatial variability in flow characteristics (Metcalf & Eddy, 2003). Residence time distributions were determined for each of the ten inner wetland sample locations on transects A, B, and C. F-curve analysis for detention time used a sigmoidal model to calculate mean detention time as a function of the portion of tracer detected at a sample location over the time of sampling, or the breakthrough curve (Metcalf & Eddy, 2003). The mean residence times were compared between the low and high marshes at each transect and along the longitudinal flowpath.

Breakthrough curve data collected at the ten sample locations were used to calculate the Morrill Dispersion Index (MDI) to compare the degree to which dispersion was occurring at different locations throughout the wetland (Metcalf & Eddy, 2003). The MDI was calculated as the ratio of the 90-th percentile (P_{90}) and the 10-th percentile (P_{10}) of the tracer breakthrough curve. To obtain these percentiles, each sample location was treated as a separate reactor where the breakthrough curve time started one time step before tracer was detected and the cumulative tracer concentration was the summation of all of the measured Br concentration at each individual site.

IV.3 Results and Discussion

Wetland Residence Time Distributions

Impulse experiment data were used to characterize wetland residence-time distribution. The average measured outflow rate during the controlled flood event in the H-flume was 3.69 m^3/min. The wetland water volume was determined to be 250 m^3 through the developed stage-volume relationship and the observed stage during spring flood event. From these measurements, nominal retention time was approximately 68 min. Mean residence time was determined to be 100 min from the concentration versus time tracer response curves created with the impulse experiment (Fig. 4.2). The effective volume ratio for this wetland during the controlled flood event was 147%. In theory, the nominal residence time is always higher than determined mean residence time due to void spaces that are present in the wetland that are created by vegetation and dead flow zones. Therefore, in theory, a wetland never attains 100% volumetric efficiency. The discrepancy here can most likely be explained by error that was incorporated in the stage-volume relationship created by using a ground survey and the cut/fill tool in AutoCAD Civil 3D. However, the information from the calculations provides useful information. In a synthesis of wetland studies, volumetric efficiencies above 100% were common in free surface wetlands (Kadlec, 2009a).

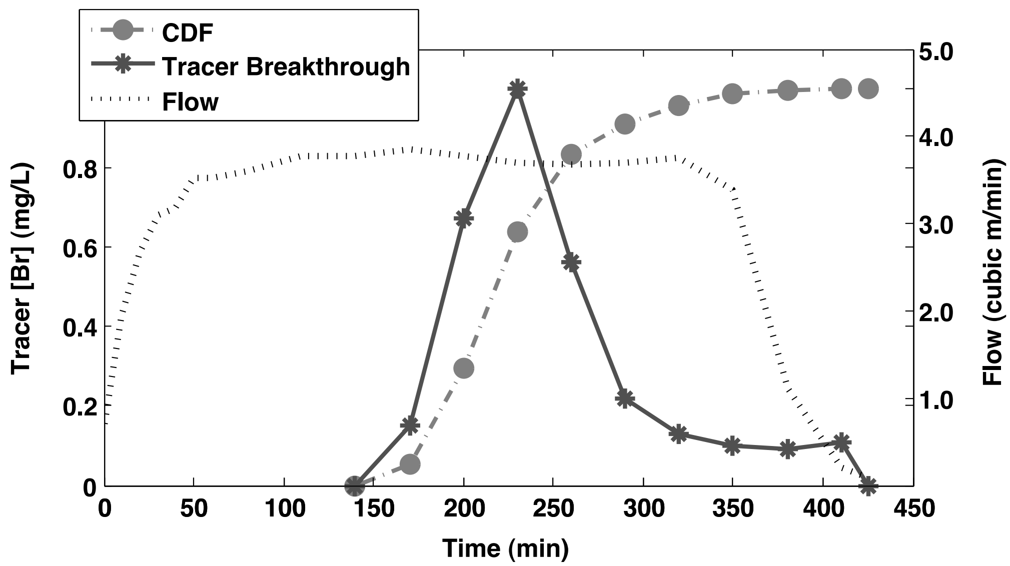

Figure IV.2 Tracer breakthrough curve and associated cumulative distribution function as produced by the impulse experiment in Spring 2009.

The variance of the tracer pulse around the mean residence time was used to calculate dispersion parameters. The wetland dispersion number was found to be 0.10, which corresponds to N = 5.3 for a TIS model. These values are not in the range of mixing that would describe intermediate amounts of axial dispersion as described by Levenspiel (1972). Therefore, this CW experiences higher mixing than is allowed by the assumptions of the dispersion models. This is common in FWS wetlands; however, the model is still a useful tool in describing the wetland mixing in the context of other literature (Kadlec and Wallace, 2008). In a synthesis of 35 tracer studies on FWS wetlands, the average N was 4.1. A study included in the syntheses on a FWS wetland with a similar average depth and surface area to that of this study's CW found an N of 10.7 (Unpublished data, Champion Paper Co., from Kadlec and Wallace, 2009).

The average superficial velocity through the wetland was estimated to be 1.2 cm/s by dividing the total length of the flowpath (70 m) by the mean residence time. While studies have shown bed drag to be negligible as compared to vegetative drag in wetlands (Nepf, 1999) and that the application of open channel flow equations should not apply to variable-depth wetlands (Kadlec, 2009a), Manning's *n* was used in this study as a tool for relative comparisons of the roughness in the high and low marshes. Based on an overall average velocity and depth over the entire wetland and assuming the wetland could be described as a wide channel, a Manning's *n* of 0.62 was determined to describe the overall roughness of the wetland.

Bromide breakthrough curves were created for each sample location by evaluating Br concentration over injection time (Fig. 4.3). F-curve analysis at each sample location determined mean residence time (τ) of water volumes at each location (Table 4.3). Generally, τ was lowest at Transect A and highest at Transect C. Also, higher residence times were measured in the high marsh at all transects as compared to their associated low marsh.

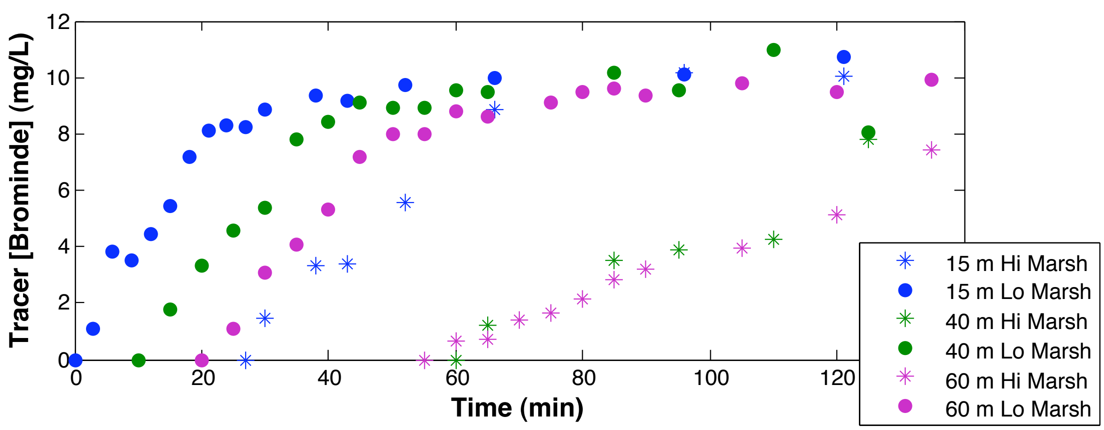

Figure IV.3 Example tracer breakthrough curves from the fall experiment at one high marsh and one low marsh sample location at each transect.

To help understand the mechanisms that created variation of retention times between the high and low marshes, Manning's *n* was determined for each sample location. In both events, a longitudinal change in Manning's *n* was observed (Table 4.2). At Transect A, Manning's *n* is lowest in the low marsh as compared to the adjacent high marsh. At Transect B, there is a transition where Manning's *n* is varied between the high and low marshes and between the high marshes between events. Finally at Transect C, Manning's *n* is lowest in the high marsh as compared to low marsh. This pattern may be a result of the pumping methods necessary for the experiments, which created turbulent flow in the confined inlet channel that then met the relatively wide channel of the wetland. The low marsh had a higher Manning's *n* than the high marsh areas at B and C with the exception of a high *n* at Location 7, where breakthrough curves suggested that this area might be affected by backwater characteristics.

Table IV.2
Flow characteristics determined for each sample location during both season events; mean residence time (τ), superficial velocity (V), flow depth (D), and Manning's *n*.

			Fall				Spring			
Transect	**Feature**	**Location**	**τ (min)**	**V (cm/s)**	**D (m)**	***n***	**τ (min)**	**V (cm/s)**	**D (m)**	***n***
A	HI	1	18.2	1.17	0.20	0.85	21.9	0.97	0.22	1.40
A	LO	2	14.9	1.54	0.19	0.64	18.0	1.27	0.21	1.06
A	HI	3	47.7	0.53	0.10	1.21	44.5	0.57	0.12	1.64
A	LO	4	12.9	1.65	0.25	0.70	22.2	0.96	0.26	1.63
B	HI	5	89.6	0.69	0.07	0.70	52.2	1.19	0.08	0.58
B	LO	6	28.5	2.02	0.29	0.63	29.9	1.93	0.30	0.87
B	HI	7	80.1	0.82	0.03	0.33	48.1	1.31	0.18	0.92
C	HI	8	67.2	1.46	0.09	0.39	48.9	2.00	0.08	0.34
C	LO	9	36.3	2.60	0.28	0.48	41.7	2.26	0.26	0.69
C	HI	10	58.4	1.67	0.06	0.27	52.8	1.84	0.05	0.28

The roughness pattern between the low and high marshes shows that the roughness of the channel in the low marsh creates more friction on flow than that of the high marsh. This roughness may be a combination of higher variability in bed topography as well as the increased complexity of the surface area of the *Sagittaria latifolia* (arrowhead) and *Pontederia cordata* (pickerelweed), which is the dominant vegetation in the low marsh, as opposed to the smaller-diameter and consistently shaped characteristics of the *Scirpus validus* (bulrush) and other grasses, which dominate the high marsh areas. Variations in roughness and velocities between the fall and spring event were possibly a function of the difference in water surface slope during the fall (Fig. 4.4) and spring (Fig 4.5.) event, as it is a sensitive parameter in the calculation of Manning's *n* and has been found to significantly influence floodplain hydraulics (Harvey et al., 2009). In summary, independently measured variables included water depth, water surface slope, and residence time (through breakthrough curve data). Dependent variables included velocity and Mannings *n*, which were calculated.

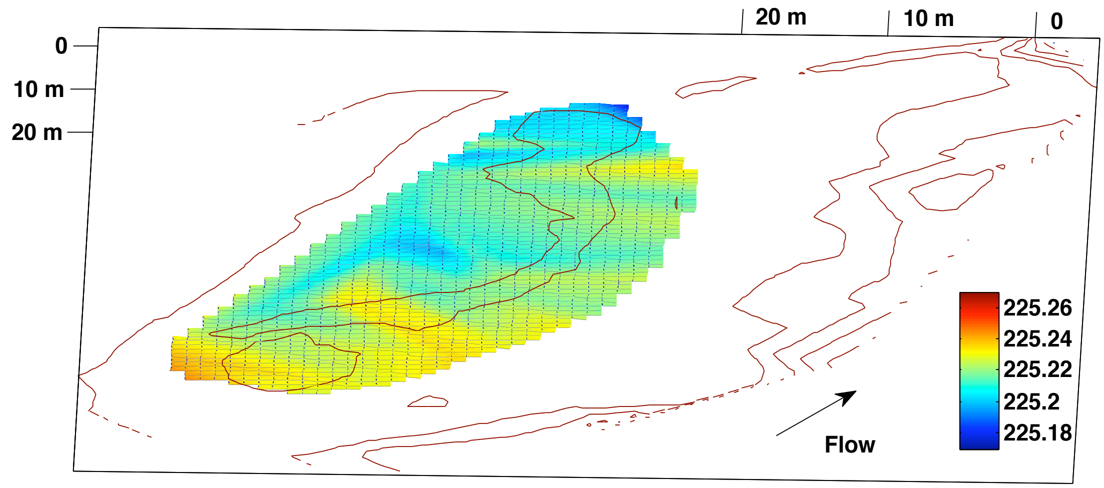

Figure IV.4 Fall event 1-m mesh water surface as delineated through field surveying of inundated area and linear interpolation.

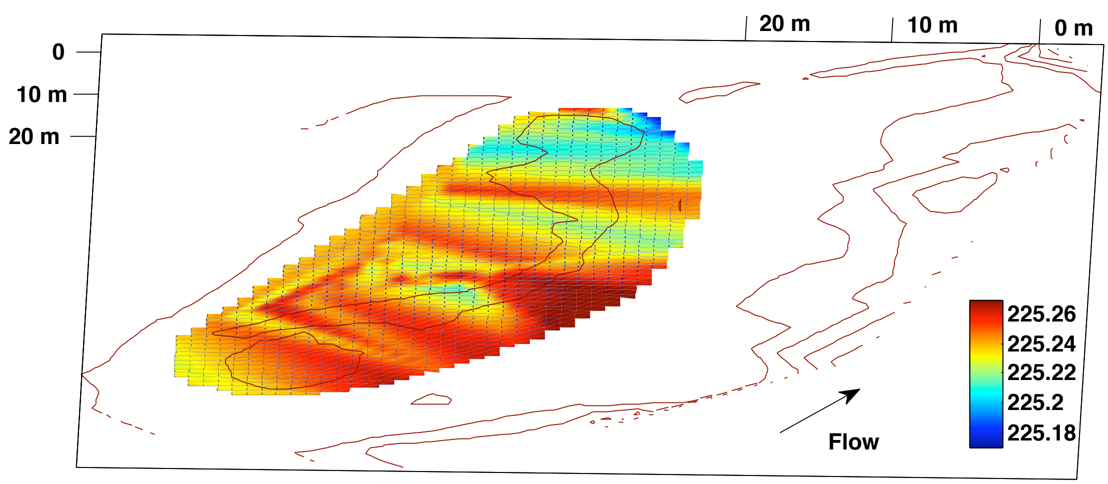

Figure IV.5 Spring event 1-m mesh water surface as delineated through field surveying of inundated area and linear interpolation.

Average Manning's *n* was determined for each season in both the high and low marshes (Table 4.3). Roughness values in the fall did not vary significantly between the high and low marsh. However, in the spring, average roughness in the low marsh in the spring was higher than the high marsh. This may be attributed to the higher leaf surface area that is characteristic of the arrowhead and pickerelweed vegetation as compared to the bulrush and grasses that dominate the high marsh. Overall, average Manning's *n* was significantly higher in the spring than in the fall (p=0.088), indicating that flow was experiencing more friction due to the spring-time characteristics than the fall.

Table IV.2
Average Manning's *n* and associated standard deviation as grouped by season and feature location.

Season/Feature	n	Average Manning's *n*	Standard Deviation
Spring/Low Marsh	4	1.06	0.41
Spring/High Marsh	6	0.86	0.57
Fall/Low Marsh	4	0.61	0.09
Fall/High Marsh	6	0.62	0.36

Macrotopography-Induced Mixing

The MDI was used to determine if the residence time distributions found at each sample location were characteristic of plug flow or complete-mixed reactor. Plug flow patterns were identified when MDI values were at or below 2 (USEPA, 1986). The MDI was highest at Transect A in both events, possibly being affected by the forebay just up-wetland (Fig. 4.8). MDI values at Transects B and C were consistently between 4 and 8. This showed that as flow moved longitudinally through the wetland, there was a threshold of degree of dispersion. Transect A spring MDI values were lower at three of the four sample locations as compared to the fall. This may have been an effect of a slightly higher flow rate in the spring that may have decreased the mixing effect off of the back end of the forebay to decrease the MDI value to levels closer to those measured at Transects B and C. At B and C, spring MDI values were higher, which shows a higher degree of dispersion in the spring event.

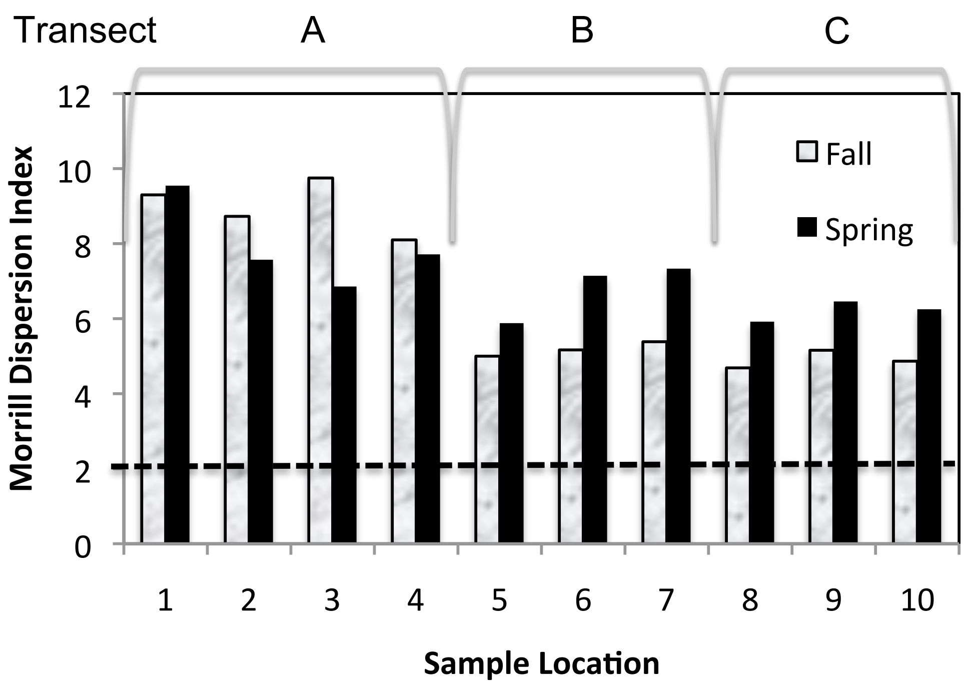

Figure IV.6 Morrill Dispersion Index values calculated from tracer breakthrough curves produced by the step input experiment as compared by season.

Flow patterns were observed in the residence time calculations from the step response tracer. A 1-m mesh grid was created using a radial interpolation method in MATLAB (The Matworks, Inc., Natick, Massachusetts) and was informed by the retention times calculated at each of the ten grab sample locations. The resultant interpolation shows the spatial distribution of retention times during the step response tracer time period. In both events, the wetland macrotopography clearly influences the residence time of the introduced flow. This is evident by the pattern of residence time in the mesh grid, which was observed in both events to exhibit lower residence times along the low marsh surrounded by areas of higher residence times in the high marsh. The fall event was characterized by a preferential flow path down the low marsh and adjacent areas of lesser mixing in the high marsh. Contrastingly, the spring event was characterized by more mixing between the low and high marsh areas, resulting in lower spatial variability of retention times measured throughout the wetland (Fig. 4.9). These findings support the results of the MDI analysis between seasons.

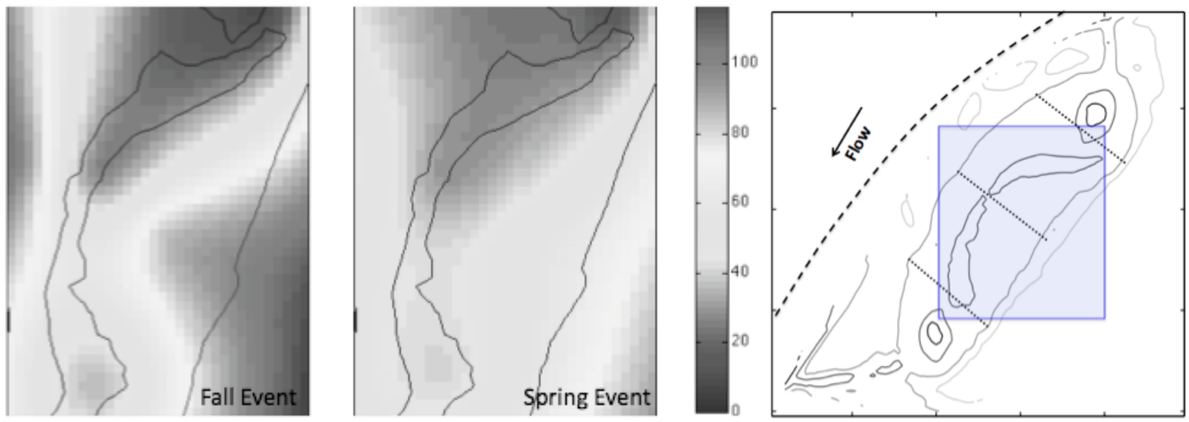

Figure IV.7 Mean residence time (min) as determined through tracer breakthrough curves in step input experiment and spatially interpolated through a 1-meter grid overlying wetland topography.

The increased mixing in the spring event may be attributed to the increased flow rate (as compared to that of the fall event) which decreases the effects of bed roughness by collapsing the boundary layer, or possibly the presence of live plants that meet flow with rigid stems and leaf surface areas, both of which influence flow patterns by inducing roughness. Furthermore, in the spring event, floating mats of algae were present at the beginning and remained in the low marsh throughout the event. This mat on the surface of the water may affect the way flow currents diffuse into the surrounding high marsh areas. Finally, these patterns may be an effect of the difference in starting water table elevations between the seasons. Vertical hydraulic gradients in the wetland would have been order of magnitudes higher in the unsaturated surface of the fall event than the saturated surface in the spring event. This may have caused flows to travel more vertically through the soil column than axially through the surface water column. These observations are most likely a result of the mentioned processes combined.

Tracer samples were collected for 135 min during the fall event, while sampling took place for 200 min in the spring event. While more flow was traced and collected in the spring event, the events achieved approximately the same level of tracer saturation at all sample locations. Furthermore, residence time determinations used the maximum bromide concentration measured at each location to normalize all previous sample concentrations, allowing each site to be analyzed independent of degree of tracer saturation.

Total Suspended Solids Removal

While previous studies have shown the collection of samples through automated sampler units to produce higher concentrations of nutrients (such as total P) than those compared with samples collected using a grab technique (Kadlec and Wallace, 2008), samples collected in this study at the wetland inlet and outlet were obtained using the same sampling technique, and therefore, would be comparable to each other while reporting removal rates. The wetland removed approximately 70% of the sediment load injected into the wetland in both events, as measured by the comparisons of the sediment flux measured at the inlet and the outlet (Fig. 4.10). The similarity in removal trends of both events indicates that the removal mechanism was not significantly affected by seasonal changes. Suspended sediment settling in FWS wetlands is a function of particle size and terminal velocity, which is reached quickly for larger particles (Kadlec and Wallace, 2008). To address the mechanism of removal, suspended sediment from the grab samples was examined as a function of residence time at each sample location. Data from both events showed that as residence time increased, suspended sediment concentration decreased (Fig. 4.11). These observations suggest that a greater consideration of mean residence time in design could result in more TSS removal with stormwater BMPs.

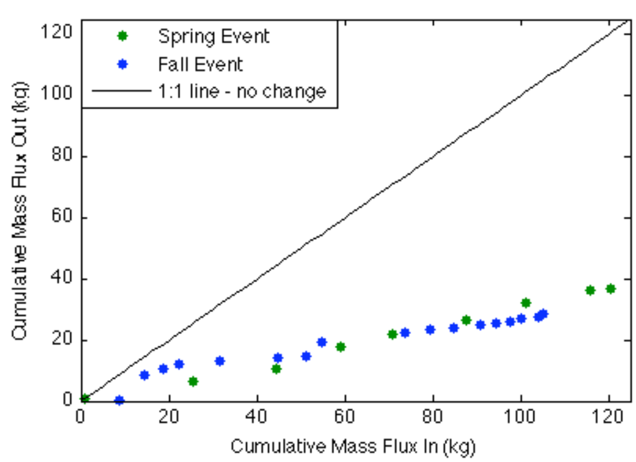

Figure IV.8 Cumulative mass flux of TSS from the inflow and outflow of the wetland during two controlled flood events.

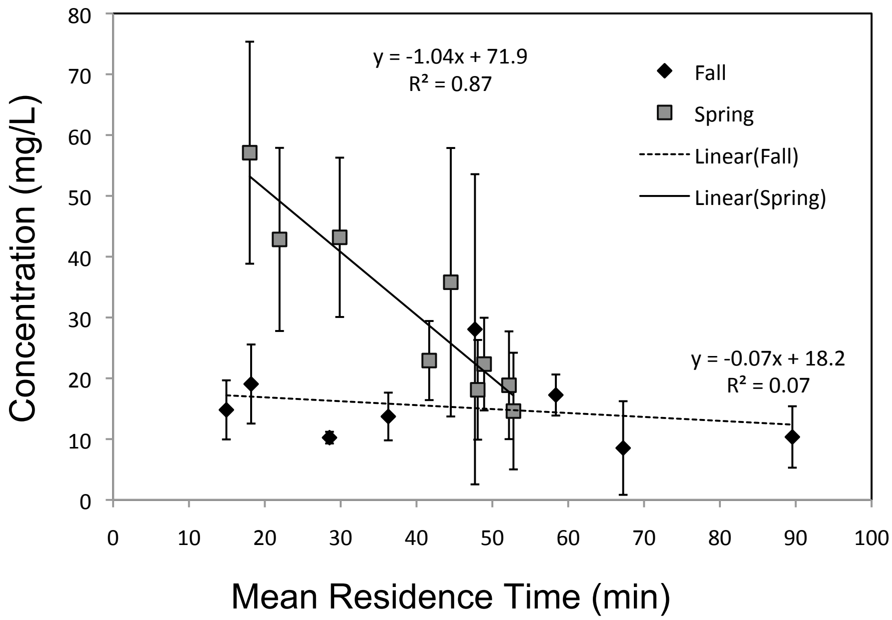

Figure IV.9 Total suspended solids at sample locations as a function of mean residence time as determined through analysis of tracer breakthrough curves in two flood events.

IV.4 Conclusions

The CW was found to be functioning with a degree of hydraulic mixing that was indicative of a well-mixed system. Mean residence times were shorter in the low marsh areas as compared to the high marsh areas. The overall roughness of the wetland was higher in the spring than in the fall, and this change was attributed to the presence of rigid vegetation in the spring and its absence in the fall. Spatial variability in roughness throughout the wetland showed that the broad-leafed vegetation species in the low marsh along with the increased bed roughness had an effect on flow velocities. The incorporation of macrotopographic features and a variety of wetland species produced spatial variability in surface depth, superficial flow velocity, mean residence time, and roughness.

Wetland macrotopography influenced the hydraulic transport of flow and associated nutrients in the constructed wetland. As residence time increased in pockets of transient storage, total suspended solids concentration decreased. This relationship was observed in both seasons, showing that the removal mechanism was not affected by temporal changes. This study shows that increasing residence time of sediment-laden flows and exchanging water in pockets of transient storage in FWS wetlands will decrease suspended solids concentrations.

Future studies in the area of flow-through wetland hydraulics should focus on long-term effects of vegetation establishment on storage and flow patterns. More information is needed to inform design considerations for integrating macro-and micro-topography in wetland designs and how it influences the development of nutrient processing mechanisms. Effective treatment wetland design must consider the hydraulic regimes needed for the capture and processing of a variety of species of non-point source pollutants and successfully integrate all of these conditions into the design process.

Acknowledgements

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V. Event-Scale Nutrient Attenuation in a Small Constructed Floodplain Wetland

V.1 Introduction

Riparian wetlands have been identified as important landscape features for the management of nutrients reaching receiving waterbodies, such as drinking water supply reservoirs and ecologically sensitive estuaries (Mitsch et al., 2001). Efforts have been made in the field of floodplain and wetland restoration to hydrologically reconnect riparian areas with impacted stream channels with the goal of recovering the ecological services provided by riparian wetlands (Acreman et al., 2003). Non-point source pollutants of nitrogen (N), phosphorus (P), and sediment have been shown to be attenuated from flows through natural and constructed floodplain wetlands in the time scale of natural storm events (Casey and Klaine, 2001; Noe and Hupp, 2007; Schulz and Peall, 2001).

The driving pollutant removal mechanisms in a treatment wetland are physical settling of solids, chemical sorption of dissolved constituents, and biological conversion by microbes and wetland vegetation (Kadlec and Wallace, 2009). The time scale at which these mechanisms occur varies dependent on seasonal factors such as temperature and solar energy inputs as well as other variables such as hydrologic and chemical inputs. Other factors have also been determined to impact treatment performance of wetlands, such as watershed area (Carleton et al., 2001), plant processes (Neubauer et al., 2005b; Reddy et al., 1999), and antecedent moisture conditions before and after storm events (Kadlec, 2010). Identifying a monitoring method and correctly quantifying nutrient attenuation by these treatment wetlands is essential for assessing their effectiveness at pollutant removal.

Phosphorus (P) is of particular concern in most inland temperate regions because it is often the limiting nutrient for primary productivity and, therefore, controls the relative amount of standing crops of aquatic autotrophs (Stevenson, 1996). These autotrophs are the primary control of dissolved oxygen levels in surface waters, which is a sensitive water quality parameter for fish and macroinvertebrates (Wilhm, 1968). Currently, there are no surface water standards, but US Environmental Protection Agency (USEPA) recommends below 0.50 mg/L TP for surface waters (USEPA, 1986). However, levels of TP in excess of 0.03 mg/L TP have been determined to produce nuisance algal blooms in temperate lotic environments (Dodds, 2002).

In a comprehensive review of literature on wetland nutrient removal, riparian wetlands were found to reduce total phosphorus (TP) loadings; however, loadings of soluble P were likely to increase rather than decrease (Fisher and Acreman, 2004). Greater TP concentrations in ground water associated with poorly drained riparian buffers suggested that wetland designs for nitrate removal might not be effective for P removal (Young and Briggs, 2008). However, the highly dynamic hydrology associated with riparian areas combined with native wetland vegetation may create favorable conditions for capture of particulate and dissolved phosphorus that may be traveling through preferential groundwater flowpaths or in overland stormflows (Braskerud, 2001; Fisher et al., 2009; Van de Moortel et al., 2009; Wetzel, 2001).

The objectives of this study were to: 1) evaluate the event-scale nutrient attenuation capacity of a constructed floodplain wetland; 2) identify spatial variability in nutrient concentrations and removal throughout the wetland; and 3) identify temporal variability in nutrient removal between the fall and spring events and within a controlled flood hydrograph. The significance of this work lies in its implications for created floodplain wetland design. The uniqueness lies in the fact that we were able to create controlled flood events through our constructed wetland, allowing us to control the flow into the wetland, inject tracer and nutrients, set up spatial sampling locations, and perform events in two very different seasons. This work aims to shed light on the ecological services provided by riparian wetlands with regard to nutrient management.

V.2 Methods

Site Description

We constructed a floodplain wetland adjacent to Opequon Creek as part of a larger USEPA and the Chesapeake Bay Foundation project to act as a demonstration best management practice and experimental wetland for stormwater management. Opequon Creek is a targeted watershed by state and federal efforts to reduce the amount of non-point source pollutants reaching the Chesapeake Bay (Mostaghimi, 2003a; Mostaghimi, 2003b). Storm event dissolved nutrient concentrations in Opequon Creek at the study site were as high as 3.12 mg/L NO_3, 0.03 mg/L NH_3, and 0.09 mg/L PO_4 throughout the project period of 2006-2009 (unpublished data, Opequon Targeted Watershed Grant). Total phosphorus concentrations as high as 0.16 mg/L TP were measured by the USGS from water samples collected at gage 01614830, just downstream from the study site (USGS, 2009). These concentrations are well above the 0.03 mg/L TP threshold for limiting algal blooms.

The wetland covers approximately 0.2 ha of converted pastureland and was designed using guidance for constructed stormwater wetlands provided by the Virginia Department of Conservation and Recreation (VADCR, 1999). A forebay is located at the head of the wetland and a sinuously low marsh winds through a high marsh. An exit pool is located just before the outlet channel (Fig. 5.1). More detail of the site may be found in Chapter 3.

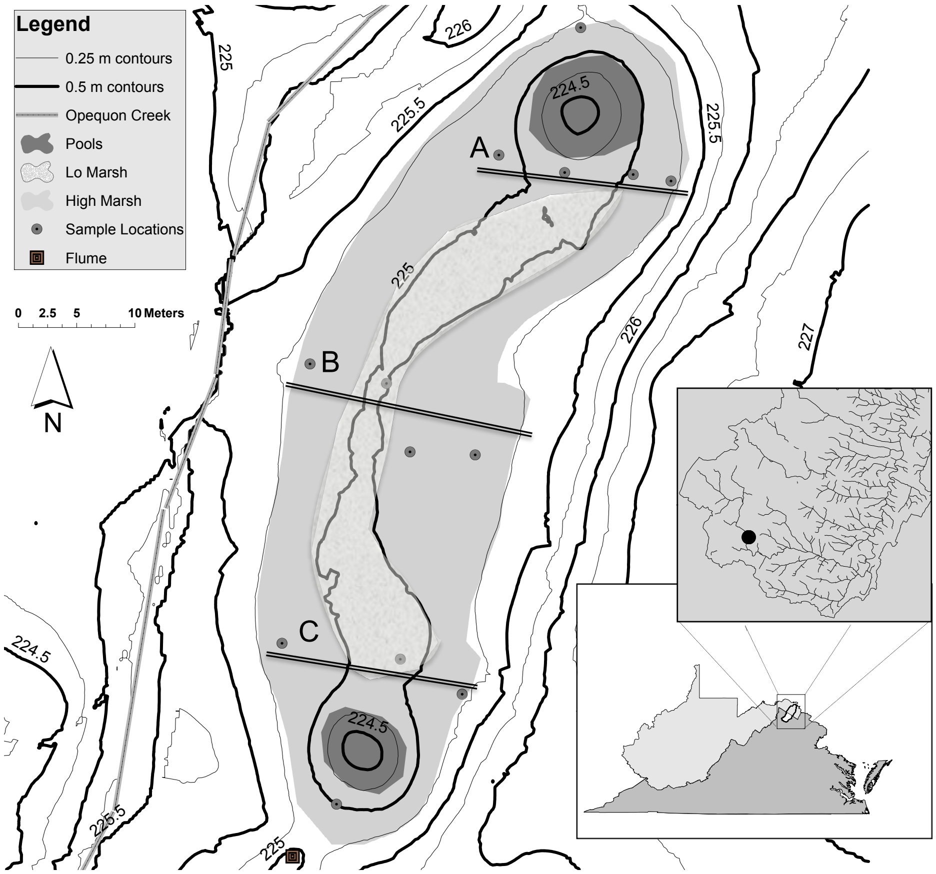

Figure V.1 Constructed wetland topography and sample locations along three flow-normal transects, near Winchester, VA, USA.

The wetland hydrologic budget was characterized as part of a parallel study in which we found the wetland water surface to be connected to the groundwater with no indications of confining layers or perching effects (Chapter 3). In addition, we found that surface storage was impacted by groundwater exchange between the adjacent hillslope and stream channel. Initial groundwater table elevations were measured in groundwater piezometers located throughout the wetland with the understanding that hydrologic conditions prior to a storm event may affect nutrient retention (Kadlec, 2010; Rucker and Schrautzer, 2010).

Controlled Flood Events

Two controlled flood events were performed at the wetland site and were intensely sampled to characterize treatment performance. The first event was in November 2008 and the second in May 2009. These time periods were selected to represent two separate seasons, fall and spring. Fall wetland conditions consisted of the presence of decaying and dormant emergent vegetation, abundance of leaf litter from perimeter trees, a low water table, minimally inundated low marsh, and considerable amounts of exposed dry soil surfaces in the high marsh. Spring wetland conditions consisted of the presence of live and rigid emergent vegetation, a floating dense algal mat in the low marsh, a high water table, complete inundation in the low marsh, and areas of inundation on the high marsh. Tracer injection experiments were coupled with nutrient sampling in each event and provided residence time data for each sampling location. Detailed tracer methodology and analysis may be found in Chapter 4.

Event Sampling

The wetland inlet and outlet were instrumented with automated sampling units called 6712 Portable Samplers (ISCOS)(Teledyne ISCO, Inc., Lincoln, NE). The inlet unit was comprised of a 750 Area Velocity Flow Module and automated sampling system, while the outlet unit utilized a 730 Bubbler Flow Module to capture stage in the 0.3-m H-flume and an automated sampling system. Inflow was quantified using continuity of flow through a known cross-section at a measured velocity, while a flume-rating curve quantified outflow based on stage measurements. Three sampling transects were installed normal to flow, spanning the width of the wetland, and allowing for sampling in low and high marsh areas of the wetland (labeled A, B, and C; Fig. 5.1). Twelve sampling locations were established along these three transects, three along each transect, and one location at the inlet and outlet. Staff gauges were installed throughout the wetland and were used to monitor storage stage over the course of the event.

Three different types of sampling techniques were used to quantify nutrient dynamics during the events. Inlet-outlet data were collected from the inlet channel and the outlet flume using a time-dependent sampling program on the ISCO unit. This sampling was continuous over the course of the entire event in the fall event. Flow was sampled and composited at time steps at the inlet and the outlet and utilized in percent removal calculations. The ISCO units malfunctioned at the beginning of the spring event, making them unreliable for the inlet-outlet sampling. Instead, grab samples in the inlet and outlet were used in place of ISCO composite samples. Snap-shot grab samples were collected at twelve locations throughout the wetland within a 5-minute window (see sample locations, Fig 5.1). This was performed four separate times, all occurring after the stage steady state was determined by field observations of consistent stage on the staff gauges. A synoptic sample was collected during the steady state of the spring event. This sampling was performed within at 20-minute window at 13 sites additional to the original 12 (for a total of 25 sites) during steady state.

Water quality parameters were measured in the wetland storage water. In the fall event, a calibrated YSI 556 Multi-parameter handheld unit (YSI, Inc., Yellow Springs, Ohio, USA) recorded point measurements of dissolved oxygen (DO), temperature, and pH at each sampling location and at a time in the middle of the flood event. In the spring event, a Hydrolab data sonde (Hach Company, Loveland, Colorado, USA) was calibrated and deployed to continuously record measurements of DO, specific conductivity, temperature, and pH in the low marsh at transect B.

Water temperature was used as an indicator of hydraulic mixing in the spring event based on the assumption that the injected stream water would have a different temperature than the relatively stagnant storage water of the wetland. A mixing indication chain was constructed of temperature probes strung along steel-linked chain and suspended within slotted PVC stilling well in the center of the exit pool located just before the outlet H-flume. Twelve probes were placed at 10-cm increments along the depth profile in the pool and logged water temperature every minute during the spring event. The time series data showed changes in temperature that informed the breakdown of event hydrographs into segments: primary storage replacement, transient storage replacement, and steady state.

Lab Analyses

Upon collection, samples were handled according to standard protocols for quality assurance (APHA, 2000). Nutrient analyses included (APHA, 2000 standard method in parentheses): total suspended solids (TSS, 2540 D), ortho-phosphorus (PO_4, 4500-P G), nitrate + nitrite (NO_3, 4500-NO_3 I), ammonia (NH_3, 4500-NH3 H), total phosphorus (TP, 4500-P H), total nitrogen (TN, 4500-N C), and non-purgeable organic carbon (DOC, 5310-D/H). The NO_3 analysis utilized NO_3-to-NO_2 conversion. Throughout this paper, reported NO_3-N is the measured NO_3+NO_2 content of the sample.

Nutrient Attenuation

The capacity of the wetland to attenuate nutrients on event-scale time periods was analyzed using several methods. Percent mass reduction (PMR) was determined for the entire event hydrograph with the following relationship between input and output (Kadlec, 2009a):

$$PMR = \sum \frac{(Q_i C_i - Q_o C_o)}{(Q_i C_i)}$$

Equation V.1

where Q is flow (L^3/T) and C is concentration (M/L^3). This relationship between inlet and outlet flux was also used to determine PMR during steady-state flow; however, the relationship was simplified under the assumption of equal inflow and outflow. The total mass removed during each event was ultimately determined as the difference in cumulative mass flux between the inlet and outlet.

Linear models were fit to time series data of cumulative mass flux of constituents at the inlet and outlet. These linear models were used to calculate nutrient attenuation rate. The slope of these linear models quantified the change in mass flux over time. Attenuation rate (M/T) was determined as the difference in the slope of the linear model of the inlet and that of the outlet. A comparison of linear models with the variables of location (inlet or outlet), event time (min), and a combination of the two was performed in the open-source statistical package R Project (R Development Core Team, Vienna, Austria). Significant differences between the inlet and outlet models were determined with a significance level of $p<0.01$.

First order rate constants (k) were determined using the following relationship between measured percent removal data and flow rate and assuming a negligible background concentration (Dortch, 1996):

$$1-\frac{RE}{100}=e^{-k_a/q}$$

Equation V.2

where RE is percent removal, q is flow rate (L^3/T), and k_a is the first-order volumetric removal rate constant (L/T).

Spatial and Temporal Variability in Attenuation

Temporal and spatial variability in nutrient concentrations and attenuation were quantified for the two events. Temperature data collected in the depth profile of the exit pool informed the delineation of segments of the hydrograph for more detailed analysis of how nutrient dynamics changed over the course of the generated hydrograph. Specifically, these data were used to delineate the segments of time when the existing storage was being pushed from the wetland and the 'first flush' was occurring and when the full volume of this deepest point of the wetland was fully mixed. A steady state analysis was assumed to begin after a time when all the temperature sensors in the chain were trending the same and reflecting the warming of the ambient air temperature.

Spatial variability of nutrient concentrations was investigated through simultaneous grab sampling at 11 sampling locations (Fig. 5.1) throughout the wetland during a total of four times in each event. Samples collected along the installed transects were compared to determine if longitudinal distance from the inlet had an impact on pollutant concentrations. Samples collected within each transect were compared to determine if location within the high marsh or low marsh impacted concentrations.

Residence Time

Cross-correlation analysis was used to determine the temporal discrepancy between the inlet and outlet PO_4 flux time series (Davis, 2002). This discrepancy is an estimate of the residence time. The outlet time series was shifted by lag intervals of 10 min until the two series' trends matched as indicated by the highest correlation coefficient (R^2). Residence time was then determined to be the lag time associated with the highest R^2.

V.3 Results and Discussion

Event Hydrology and Nutrient Loading

Average flow and nutrient loading rates were higher in the spring event as compared to the fall event (Table 5.1). Inlet concentrations of PO_4 -P, TP, NH_3 -N, DOC, and TSS were higher in the spring event as well (Table 5.2). Overall, the total volume treated by both events was similar; however, the starting water table elevation in the fall event was considerably lower than in the spring event, resulting in the larger storage volume retained in the fall event. Average dissolved oxygen levels were higher in the fall, as expected with the lower water temperature.

Table V.1
Pertinent characteristics of the fall and spring controlled flood events on the constructed wetland.

	Fall	Spring
Volume Treated (thousand L)	1950	1900
Event Time (H:MM)	8:12	7:18
Avg Q (m^3/min)	42.1	49.4
Storage (m^3)	274	233
Starting Head (m)	-0.8	-0.05
Average Air Temp °C	5.1	10.6
Average Water Temp °C	9.5	16
Average DO	10.9	9.5
Average pH	8	8

Table V.2
Average constituent concentrations at the wetland inlet and outlet during the fall and spring controlled pumping experiments (range in parentheses).

		PO_4-P µg/L	NH_3 -N µg/L	NO_3 -N mg/L	DOC mg/L	TP µg/L	TN mg/L	TSS mg/L
Fall	**In**	4.9 (1-11)	25.3 (9-52)	2.8 (2.7-3.0)	1.5 (1.1-2.5)	60.8 (33-217)	3.2 (3.0-4.0)	54.5 (10-147)
	Out	4.7 (2-9)	14.6 (9-28)	2.6 (1.7-2.9)	2.3 (1.4-8.2)	62.6 (22-223)	3.1 (2.7-3.2)	33.6 (6-214)
Spring	**In**	13.5 (4-25)	38.5 (24-49)	2.2 (1.7-2.4)	1.5 (1.2-2.1)	65.2 (36-93)	3.1 (1.9-3.6)	67.0 (36-135)
	Out	12.8 (7-19)	17.5 (11-30)	2.1 (0.2-2.3)	2.5 (1.4-9.4)	54.5 (34-90)	2.5 (0.7-3.1)	24.8 (6-67)

The chemical fractionation of nitrogen and phosphorus species in the inflow and outflow varied between the fall and spring events (Fig. 5.2). Most of the N entering and leaving the wetland was in the form of nitrates; particulate N comprised more of a percentage of the inflow N in the spring (29%) relative to the fall (12%). Particulate P was the most abundant P form in the inflow and outflow; PO_4 -P comprised a higher percentage of the inflow P in the spring (21%) than in the fall (8%). Little change occurred in the relative N and P species abundance between the inlet and outlet in the fall. Changes in nitrates, particulate N, orthophosphate, and particulate P were seen in the spring event. These changes resulted in a decrease of the relative abundance of the particulate species, which suggests that physical settling was a dominant removal mechanism in the spring.

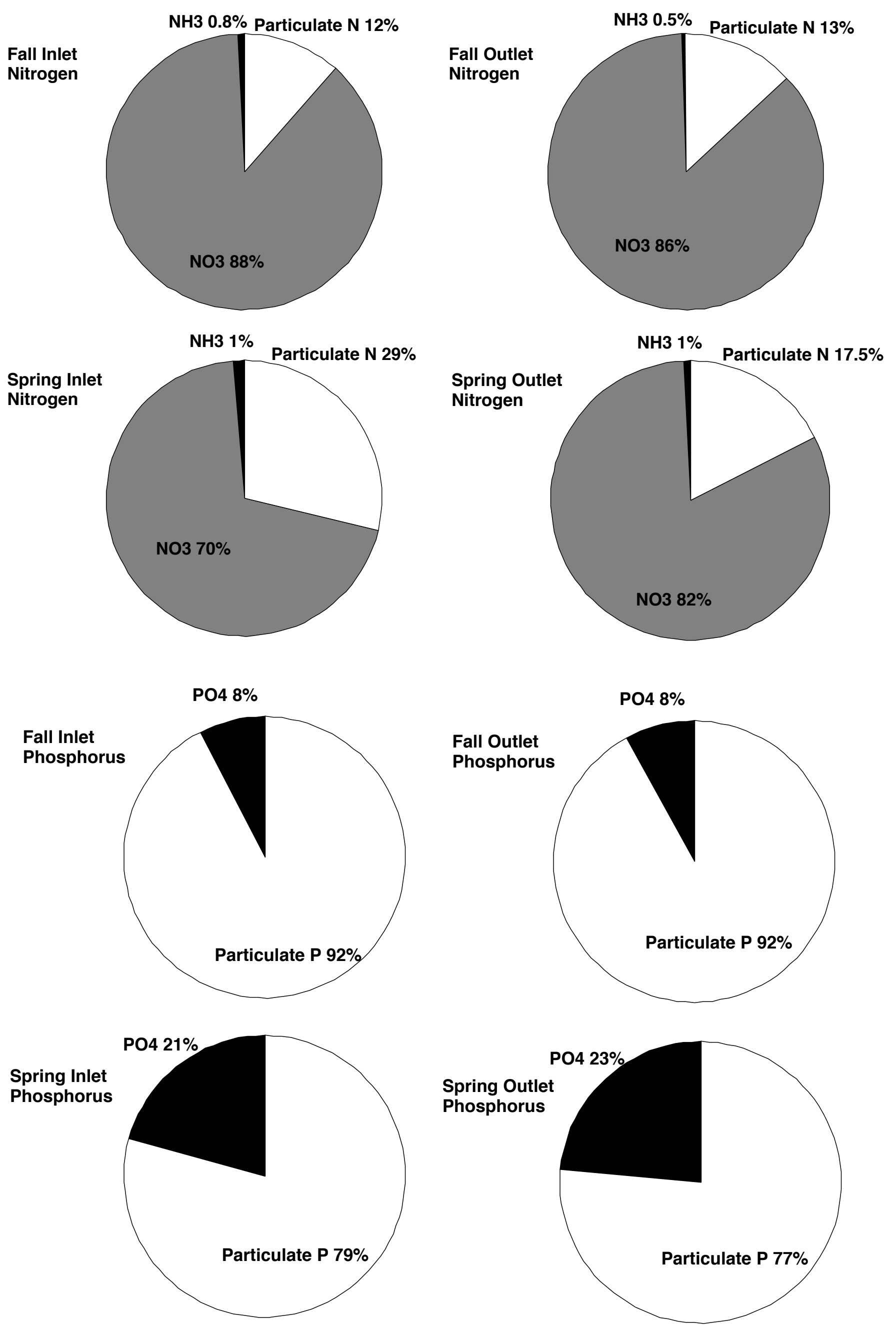

Figure V.2 Nitrogen and phosphorus fractions in inlet and outlet flows during fall and spring controlled flood events.

Nutrient Attenuation

Percent mass reduction was determined from the inlet-outlet data in both seasons for targeted non-point source pollutants (Fig. 5.3). Greater removal in the fall was measured for the constituents of PO_4, TP, NO_3, and TSS. Conversely, greater removal of NH_3 and TN was found in the spring. These removal numbers are a function of event inlet concentrations. Thus, in seasonal comparisons, inlet concentrations must be considered when comparing the events, as a higher percent mass reduction. Average inlet concentrations were higher in the spring event for all constituents except TN and NO_3.

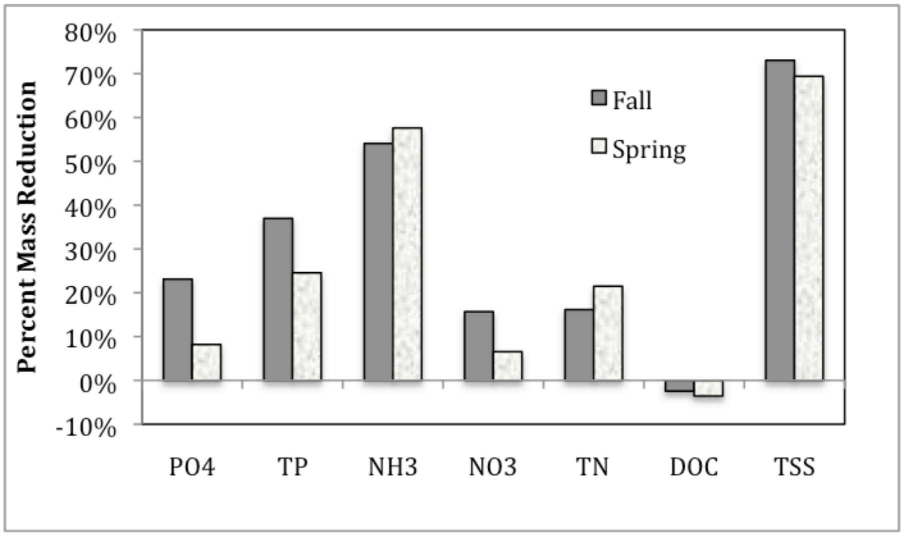

Figure V.3 Percent mass removal of targeted nutrient constituents and suspended solids from controlled flood event experiments on the constructed floodplain wetland in the fall and spring seasons.

Removal rates were higher in the spring event for all constituents (Table 5.3). While the events treated approximately the same amount of volume, the spring event was shorter in time and removed the same or more mass of constituents than the fall event. There was a two-fold increase in removal rate in the spring for the constituents of PO_4, NH_3, and TN. Since the inlet-outlet sampling was performed with two different methods (ISCOs in the fall and grab sampling in the spring), this may be a source of slight error when comparing the two events. However, the sampling protocol was consistent within the event.

Table V.3
Constituent percent mass removals, total mass removed, and removal rates (determined through linear regression) in the fall and spring controlled flood events.

Constituent	Fall % Removal	Spring % Removal
PO_4	23.1	8.2
TP	37.0	24.5
NH_3	54.1	57.6
NO_3	15.7	6.6
TN	16.2	21.5
DOC	-2.5	-3.6
TSS	73.0	69.4
Constituent	**Mass removed**	**Mass removed**
PO_4 -P (g)	2.2	2.0
TP (kg)	0.04	0.03
NH_3 -N (g)	26.0	41.4
NO_3 -N (kg)	0.86	0.26
TN (kg)	1.00	1.27
DOC (kg)	-0.07	-0.10
TSS (kg)	76.8	83.7
Constituent	**Rate**	**Rate**
PO_4 -P (mg/min)	3.0	10
TP (g/min)	56	73
NH_3 -N (mg/min)	43	109
NO_3 -N (g/min)	0.9	1.0
TN (g/min)	1.0	3.4
DOC (g/min)*	0.6	1.8
TSS (g/min)	190	200

* Rate of export.

Primary removal mechanisms may be deduced through scrutiny of nutrient data. Larger mass amounts of TP were removed relative to PO_4, suggesting that settling of particulate and particulate-bound P was acting as the primary P removal mechanism. A larger percent of the NH_3 input was removed compared to that of NO_3 input. Microbes select NH_3 over NO_3 as an N source for microbial processes, and this may be reflected in percent removals. The presence of larger amounts of DOC in the water column during the spring event may have provided heterotrophic microbial communities with a sufficient carbon source to support the utilization of available N and P, leading to faster removal rates. Furthermore, NH_3 from the transitional and anaerobic hyporheic zones may have gone through rapid nitrification to NO_3 with the transition from anaerobic conditions to aerobic conditions as DO increased in the water column by the addition of the turbulent inflow (Cirmo and McDonnell, 1997). TSS and PO_4 removal was higher in the fall event when chemical sorption and physical settling would prevail as dominant removal mechanisms over the temporally sensitive biological removal mechanisms.

A dense algal mat was present in the low marsh and sparse between emergent vegetation in the spring. Dissolved nutrient dynamics were most likely affected by the presence of this mat; however, the interactions between the algae mat community and the readily available dissolved nutrients are complex (Kadlec and Wallace, 2009). A boundary layer exists between the moving flow of the surface water and that of the pore water in wetland substrates. In spring, a high water table and a saturated high marsh may have limited the amount of PO_4 removal due to diffusion into pore water and exposure of sorption substrate surfaces to PO_4 carried by the overlying flow. The effects of this raised boundary layer may be reflected in the lower percent removal of PO_4 in the spring event.

Higher removal rates in the spring may also be attributed to the presence of active vegetation (unlike the fall when vegetation was approaching a dormant state), or the increase in ambient temperatures (Table 5.1), which affects the rates of microbial respiration and metabolism. Additionally, the increase in removal in the spring may be attributed to the increase in mixing throughout the wetland surface area (See Chapter 4). More of the nutrients that entered the wetland diffused into the transient storage zones, attenuating them in storage instead of passing them through the wetland in primary flow paths.

Another factor affecting removal may be the fact that the fall event was the first time flow-through conditions had occurred since wetland construction. The impact of this first flush may be quantified by looking at the difference in the pollutant speciation in the first samples collected from the outlet in each event. Relative to the spring event, the fall event had higher concentrations of TP, TN, and TSS as well as NO_3-N. TSS in the first fall outlet sample was more than double the concentration measured in the spring first outlet sample. Higher concentrations of the dissolved nutrient species of PO_4-P, NH_3-N, and DOC were measured in the first outlet samples in the spring event. Considering the low water table elevations in the fall and the high water table in the spring, this is evidence that in the fall event, water ran over the surface of the wetland and carried particulates through the outlet during the first flush, and in the spring event, storage water with high dissolved pollutants and low particulate concentrations was being pushed out of the wetland.

First order volumetric rate constants for TP were 6.1 m/yr in the fall and 4.1 m/yr in the spring (Table 5.4). In a review of data from 282 wetlands with inlet TP concentrations ranging from 0.007 to 126 mg/L with a median of 4.66 mg/L, Kadlec and Wallace (2009) found the median rate constant (k_a) to be 10 m/yr and load removal of 6 g/m^2-yr. In a review of free-water surface stormwater treatment wetlands in the US, wide ranges of removal rate constants for TP, NH_3, and NO_3 were reported (Carleton et al., 2001). Rate constants for these constituents found in this study were consistent with those reported in the review for wetlands of similar area and volume.

Table V.4
First order volumetric removal rate constants for the fall and spring controlled flood events determined from event time scale percent mass removal.

Constituent	Fall k_a (m/yr)	Spring k_a (m/yr)
PO_4-P	3.8	1.3
TP	6.1	4.0
NH_3-N	8.9	9.5
NO_3-N	2.6	1.1
TN	2.7	3.5
DOC	-0.4	-0.6
TSS	12	11

Attenuation Variability

Water temperature patterns measured in the exit pool were used to break down the event into segments that were considered during the analysis of the time series nutrient data. Steady state was determined to be the point at which the water temperature was consistent throughout the depth profile of the exit pool and reflected warming air temperatures (Fig. 5.4). This time was at approximately 72 minutes into the event. Inlet and outlet samples were paired by time and steady-state PMR was calculated for each time step of sampling following 72 minutes.

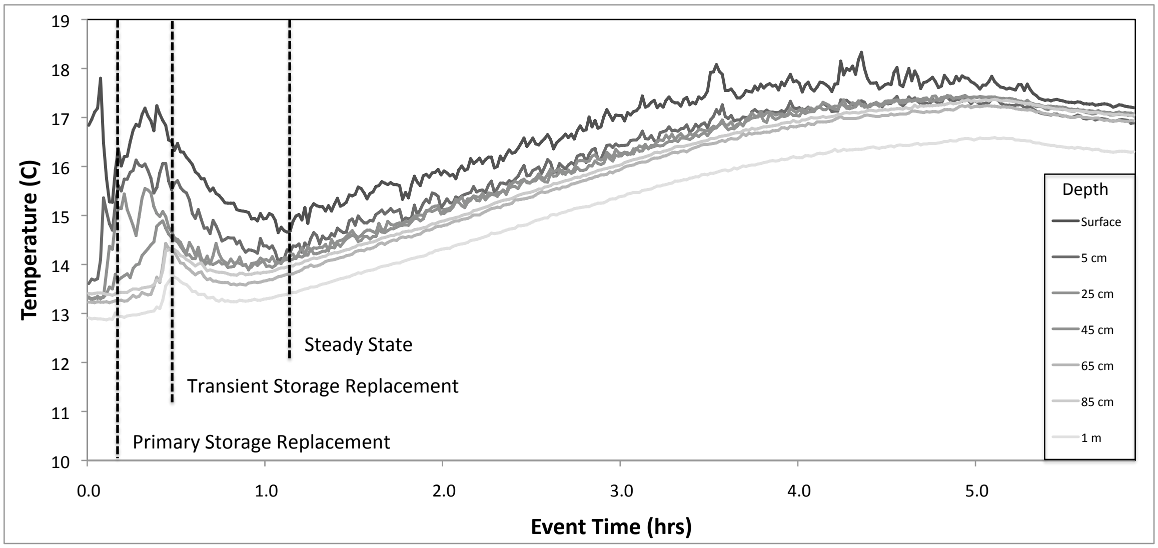

Figure V.4 Temperature measurements throughout the spring controlled flood event as recorded from a depth profile chain in the exit pool and delineations of event segments.

There was considerable variability in steady-state PMR (Fig. 5.5). Excluding outliers, the most variability was observed in percent removals of P and carbon species and generally in the fall event, where values ranged from -90% to over 100%. More variability may have been indicative of less stable mechanisms of removal dominating N removal and in the fall versus the spring event. Microbial assimilation of nutrients during the spring likely acted as a consistent removal mechanism. Such variability in PMR over a steady state event time period suggests that treatment performance measurements are highly sensitive to when samples are collected during an event and that comprehensive performance assessment must characterize complete event nutrient mass balances.

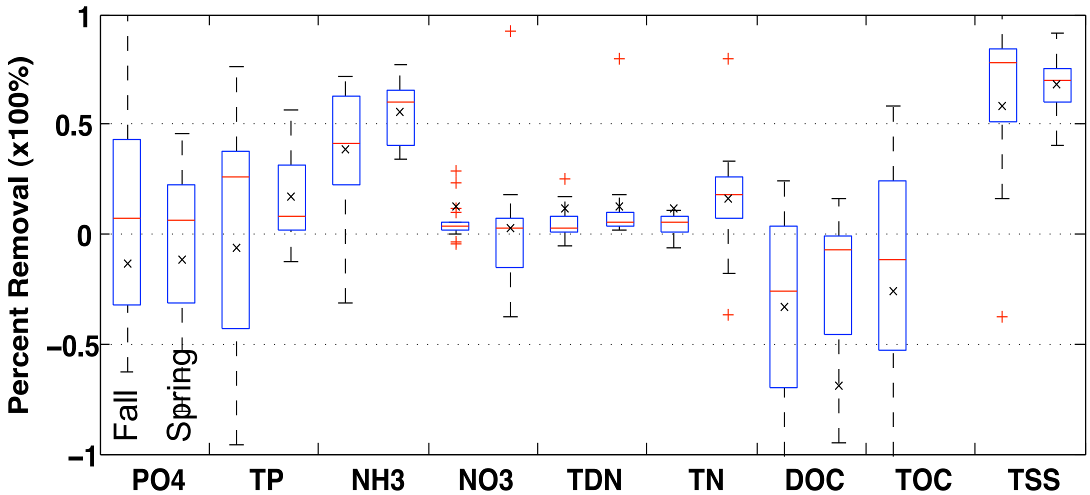

Figure V.5 Distribution of percent removal after steady state flow was established during fall and spring controlled flood events.

To further investigate variability in nutrient removal, PMR was calculated again to incorporate an adjustment for the residence time of flow in the wetland in an attempt at tracing a volume of water through the wetland to better estimate removal due to internal wetland processes occurring during steady state flow. Steady stage inlet and outlet data were paired with an offset of at least 100 min to account for residence time. This analysis produced similar PMR values, but with smaller amounts of variability relative to the first PMR determination (Fig. 5.6). The method of incorporation of a time offset equal to estimate residence time of flow may have removed some of effects of inlet concentration variability from the PR results.

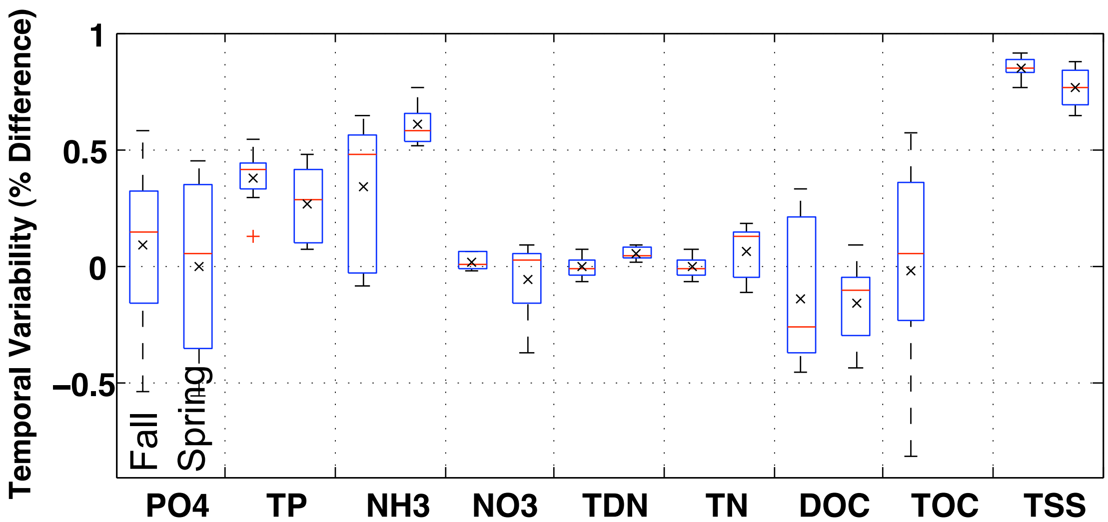

Figure V.6 Temporal variability over the event time period as determined by percent removal calculations between inlet and outlet data with a residence time lag between sample pairs.

Spatial variability in dissolved nutrient concentrations and TSS was described by the standard deviation of the four snap-shot samples and how that compared to average concentrations (Table 5.5). Sample variability was as much as 46% average wetland concentrations for nutrient constituents while TSS varied drastically in the fall event to over 400% average wetland concentrations. NO_3 concentrations were higher and less variable than those of PO_4 and NH_3. Such relatively large amounts of spatial variability in nutrient concentrations through the wetland suggest that the nutrient attenuation performance of the wetland is a result of many spatially variable processes occurring in a heterogeneous environment.

Table V.5
Percent variability in snap-shot samples during fall and spring controlled flood events, reported as standard deviation as a percent of average constituent concentration (n=12).

Constituent	Fall	Spring
PO_4 -P	46.2%	27.8%
NH_3 -N	31.5%	33.7%
NO_3 -N	12.0%	13.7%
TSS	401%	57.5%

Spatial variability was also captured in synoptic samples taken during steady state flow in the spring event. A 1-meter mesh grid was created to interpolate between PO_4 concentration measurements from grab samples taken at 25 locations throughout the wetland (Fig. 5.7). This spatial interpolation shows at this moment in time during the spring event, PO_4 -P concentrations were relatively consistent throughout much of the wetland. However, variability is seen along the perimeter of the wetland where PO_4-P concentrations tend to be greater.

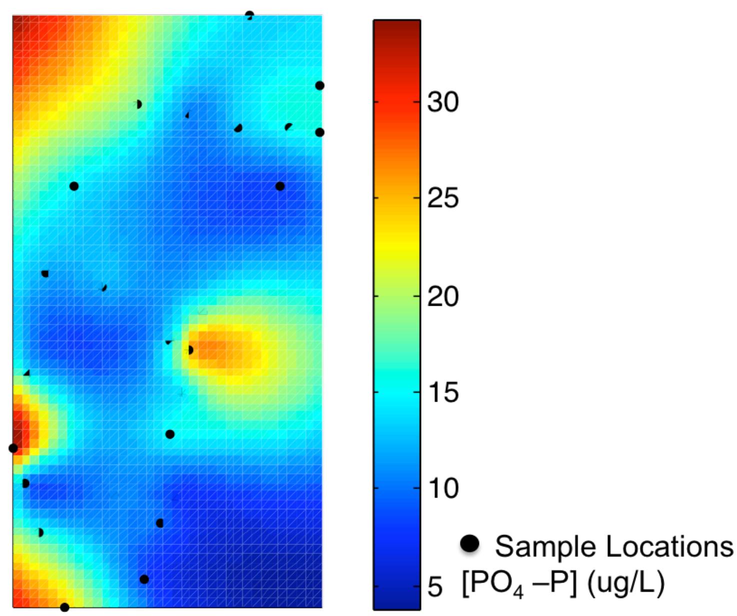

Figure V.7 Spatial interpolation of PO_4 concentrations measured at 25 locations throughout the wetland during steady state flow.

Residence Time

Residence time was determined for each event through cross-correlation analysis of PO_4 time series data collected at the inlet and the outlet. Based on this analysis, fall residence time was approximately 110 min and spring residence time was 130 min. Correlation coefficients were plotted against total analysis lag time and the curves were compared (Fig. 5.8). The spring curve is arced and has a definitive maxima while the fall curve is flatter with a less definitive maxima. This suggests that the wetland storage water in the wetland the spring before the event was pushed out by the inflow and that the two volumes had different characteristics. In the fall, the outflow was similar to the inflow throughout the sample collection.

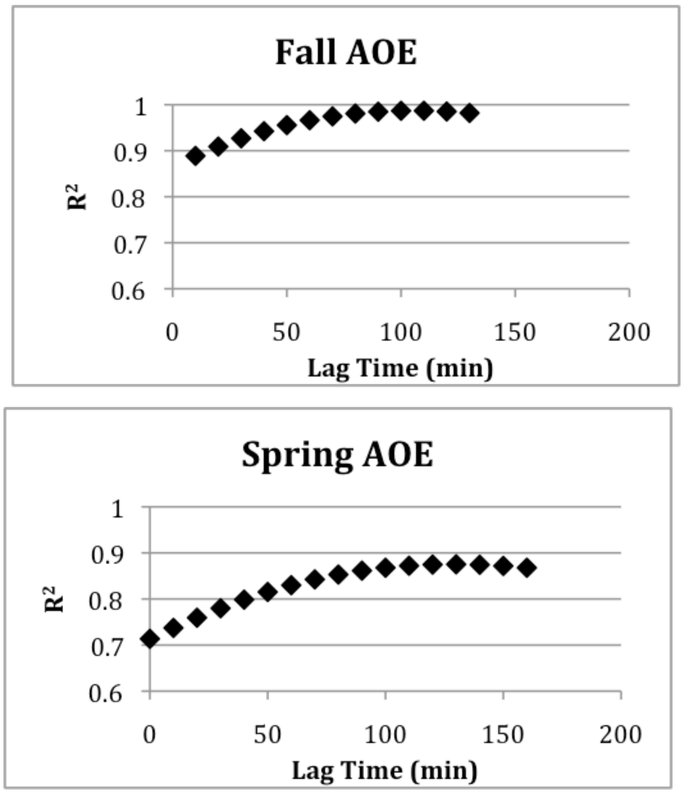

Figure V.8 Distribution of correlation coefficients as a function of induced lag time during cross-correlation analysis for determination of residence time.

V.4 Conclusions

Nutrient attenuation was measured on event time scales at the constructed wetland site. The most attenuation based on percent removed and total mass removed was in TSS. Nitrification and microbial processes may have lead to the highest percent of nutrient removal seen in ammonia. Larger removal of TP than PO_4 suggests that particulate and settling of sorbed P was the driving removal mechanism in P removal. Faster attenuation rates in the spring resulted in larger mass removal than in the fall. Warmer temperatures combined with live plants and sufficient storage DOC concentrations may have stimulated autotrophic and heterotrophic microbial communities, resulting in the attenuation of dissolved N and P from the water column.

Residence times were determined through cross-correlation of inlet and outlet data and found to be between 110 and 130 min. This was consistent with tracer injection experiments performed in parallel to these pollutant attenuation studies (See Chapter 4). Further studies on this site and other constructed wetland sites would be of interest to link residence time of stormwater and attenuation rates of pollutants.

Considering the cost and the maintenance effort required to install and ensure treatment performance of this BMP, watershed managers need to weigh the costs of this practice with the relatively small amounts of pollutant mass removed and the small window of operation of this BMP for surface stormwater management. Adaptations in the design and monitoring of this particular site may increase the potential mass removal by diverting farmland and residential runoff from small (non-overbank) events into the wetland or by investigating the potential for treatment of groundwater. This BMP may also be more effective at flood pulse and pollutant retention at a larger scale, where more land area is available and frequent flows access the BMP.

These findings show that constructed floodplain wetlands may have a role in stormwater nutrient and sediment management. This BMP has capacity to reduce nutrients and sediment in overbank flows during the event time-scale. Integrating this practice in the right locations in the landscape and as part of a larger watershed management plan would potentially decrease the amounts of non-point source pollutants discharged by rivers into receiving waterbodies.

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VI. Conclusions

In the face of growing populations in concentrated urban centers, increasingly high demand for food, and a changing global climate, the need to protect and restore the quality of our nation's waters is strong. An engineered solution to pollution control like constructed wetlands has become commonplace in practice, but little is known on the effectiveness and longevity of these practices. These research findings support the concept that allowing nutrient-laden stormwater access to ecologically enhanced areas may reduce the pollutant load to downstream receiving reaches. This research was also a test of self-design in that, with minimal land work and artificial inputs, ecological function may be restored to an anthropogenically-altered site. Furthermore, it adds to the literature on the effectiveness and variability in nutrient removal performance of engineered nutrient management practices.

Specific Conclusions

These research findings led to the following conclusions:

- The design of created wetlands without adequate knowledge of the groundwater hydrology on a site may lead to an unsuccessful wetland creation if the goals include recreating wetland function, for the hydroperiod may not suit the planted vegetation nor allow for flood storage of a magnitude as intended by the design.
- Seasonality plays a large role in the hydraulics of flow-through wetlands, and vegetation may be the largest control on how a stormflow passes through a wetland and the associated residence time distribution of pollutant-laden water.
- Spatial variability due to patchiness of vegetation and soil profile characteristics creates variability in the fate and transport of pollutants, and this variability is of a magnitude that creates a barrier to understanding low-level nutrient cycling in relatively small areas.
- Reconnecting the channel flow with its floodplain creates the potential for assimilation of sediment and nutrients from stormwater on the event time scale;

this is an ecological service that does not exist if the two are disconnected due to incision or restriction of flow.

In addition to these conclusions, we found that accounting for mean residence time in the inlet-outlet comparisons created less variability in percent removal calculation results.

Recommendations for Future Research in Constructed Wetlands

Ecological restoration is an area of research that is vital for the sustainability of our nation's water resources. The characteristics that guide the design of these restoration efforts are specific to those environmental factors unique to their ecoregion. However, the need for understanding the hydrology of the system selected for restoration or creation is universal. Constructed wetlands have a potentially critical role to play in floodplain and river restoration. In disturbed watersheds, flashy hydrology due to the increased impervious land cover causes channel morphology to change and incision to occur. Thus, stormflows that had previously been able to access the floodplains and natural filtering services found in floodplains can no longer do so, transferring pollutant-laden stormwater downstream. In the appropriate HGM setting, reconnecting channel flow with the floodplain to allow stormflows access to riparian wetlands may improve water quality.

Work is needed in the area of the longevity of built environments for water quality improvement and design optimization. As more constructed BMPs are being installed, research efforts should focus on the establishment and long-term effectiveness at nutrient removal of these practices. Potential areas for research contributions include long-term fate of pollutants that enter BMPs, inundation frequency effects on treatment performance, identification of optimal growth conditions for desirable microbes, and influence of vegetation grow-out on nutrient removal and flood-pulse retention.

Research is also needed to provide methodology for selecting optimal placement of created wetlands. With new technologies that allow us to delineate the topography of river channels and floodplains in ever more detail, spatial analysis of flow network topography may better inform wetland creation efforts in areas that are more likely to be inundated, therefore producing more treatment services per creation dollar spent. Furthermore, decades of research on wetland processes have revealed biogeochemical indicators of the fates of nonpoint source pollutants in wetlands. Sampling and precisely measuring these indicator processes coupled with detailed spatial topographic characterization will provide created wetland designers essential data necessary to select optimal sites.

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VII. Appendices

Appendix A - Bulk density data. Cores taken from the high and low marsh of Hedgebrook constructed wetland on October 30, 2008.

Location	Sample	Tin	Initial Wt (g)	Final Wt (g)	Soil Wt (g)	Water Wt (g)	Bulk Density (g/cm^3)
hi	3	1	192.06	149.04	121.86	43.02	1.24
hi	7	55	206.71	162.18	135	44.53	1.38
hi	2	20	200.86	162.51	135.33	38.35	1.38
hi	9	38	213.82	179.05	151.87	34.77	1.55
hi	8	30	221.64	180.25	153.07	41.39	1.56
hi	1	34	222.33	189.73	162.55	32.6	1.66
lo	6	21	199.46	148.7	121.52	50.76	1.24
lo	5	23	215.66	173.49	146.31	42.17	1.49
lo	4	47	221.38	177.4	150.22	43.98	1.53
						Avg	**1.45**
						Stdev	**0.15**

Appendix B - Soil sample data. Soil cores taken from wetland surface before fall controlled flood events; samples processed at the Virginia Tech Soil Testing Lab.

	Sample Loc	***pH***	***P***	***K***	***Ca***	***Mg***	***Zn***	***Mn***	***Cu***	***Fe***	***B***	***CEC***	***Ca Sat***	***Mg Sat***	***K Sat***	***OM***
								Pre-Event Samples								
Top 5 cm	1	8.28	2	27	5167	249	0.1	1	0.1	0.1	0.1	27.9	92.4	7.3	0.2	1.8
	2	8.18	2	14	5227	294	0.1	0.7	0.1	0.1	0.1	28.5	91.4	8.5	0.1	1.6
	3	8.22	2	16	5088	250	0.1	0.8	0.1	0.1	0.2	27.5	92.4	7.5	0.1	1.8
	4	8.47	2	9	5829	132	0.1	3	0.1	0.1	0.1	30.2	96.3	3.6	0.1	2.3
	5	8.5	2	9	5728	173	0.1	3.8	0.1	0.1	0.1	30	95.2	4.7	0.1	1.8
	6	8.46	2	4	5705	173	0.1	2.8	0.1	0.1	0.1	29.9	95.2	4.8	0	3
	7	8.37	2	12	5714	144	0.1	1.5	0.1	0.1	0.1	29.7	95.9	4	0.1	2.8
	8	8.6	2	6	5849	78	0.1	2.9	0.1	0.1	0.1	29.8	97.8	2.2	0	1.3
	9	8.47	2	13	5904	165	0.1	3.9	0.1	0.1	0.1	30.8	95.5	4.4	0.1	2
	10	8.43	2	10	5882	143	0.1	2.7	0.1	0.1	0.1	30.5	96.1	3.8	0.1	2.5
	11	8.14	2	45	5848	156	0.1	1.4	0.1	0.3	0.1	30.6	95.4	4.2	0.4	2.3
	14	8.35	2	15	5682	116	0.1	2.9	0.1	0.2	0.1	29.3	96.6	3.3	0.1	2
Bottom 10 cm	1	8.15	2	47	5549	196	0.1	1.3	0.1	0.2	0.1	29.4	94.1	5.5	0.4	3.4
	2	8.2	2	31	5565	243	0.1	1.1	0.1	0.2	0.1	29.8	93	6.7	0.3	2.5
	3	8.34	2	27	5556	173	0.2	2.3	0.1	0.6	0.2	29.2	94.9	4.9	0.2	3.1
	4	8.51	2	31	5751	210	0.1	2.8	0.1	0.1	0.1	30.5	94.1	5.7	0.3	2.6
	4	8.39	2	28	5756	165	0.1	1.9	0.1	0.3	0.1	30.1	95.3	4.5	0.2	3.1
	5	8.38	2	35	5789	137	0.1	3.6	0.1	0.1	0.1	30.1	96	3.7	0.3	2.1
	6	8.43	2	17	5680	187	0.1	2	0.1	0.1	0.1	29.9	94.7	5.1	0.1	3.6
	7	8.28	2	19	5664	140	0.1	1.8	0.1	0.1	0.1	29.4	95.9	3.9	0.2	4.4
	8	8.43	2	40	5875	160	0.1	2.8	0.1	0.1	0.1	30.7	95.4	4.3	0.3	2.9
	9	8.27	2	34	5827	134	0.1	3.6	0.1	0.1	0.1	30.2	96.1	3.6	0.3	2.6
	10	8.45	2	17	5577	198	0.1	2.6	0.1	0.1	0.1	29.5	94.3	5.5	0.1	2.5
	11	8.31	2	51	5830	129	0.1	1.4	0.1	0.1	0.2	30.3	96.1	3.5	0.4	3.4

Appendix C - Soil sample data. Soil cores taken from wetland surface after fall controlled flood events; samples processed at the Virginia Tech Soil Testing Lab.

	Sample Loc	*pH*	*P*	*K*	*Ca*	*Mg*	*Zn*	*Mn*	*Cu*	*Fe*	*B*	*CEC*	*Ca Sat*	*Mg Sat*	*K Sat*	*OM*
Post-Event Samples																
Top 5 cm	1	8.27	2	25	5267	228	0.1	1.2	0.1	0.2	0.1	28.2	93.1	6.7	0.2	1.7
	2	8.29	2	16	5295	279	0.1	1	0.1	0.1	0.1	28.7	91.9	8	0.1	1.6
	3	8.27	2	10	5664	171	0.1	1.1	0.1	0.1	0.1	29.7	95.2	4.7	0.1	2.4
	4	8.51	2	7	5762	106	0.1	2.9	0.1	0.4	0.1	29.6	97	2.9	0.1	2.2
	5	8.43	2	13	5964	95	0.1	4.6	0.1	0.1	0.1	30.6	97.3	2.5	0.1	1.8
	6	8.4	2	5	5703	188	0.1	2.5	0.1	0.1	0.1	30	94.8	5.1	0	3.1
	7	8.42	2	16	5683	127	0.1	1.9	0.1	0.3	0.1	29.4	96.3	3.5	0.1	2.9
	8	8.59	2	14	5834	98	0.1	2.9	0.1	0.3	0.1	29.9	97.2	2.7	0.1	1.6
	9	8.44	2	21	5785	106	0.1	3.5	0.1	0.1	0.1	29.8	96.9	2.9	0.2	2
	10	8.47	2	8	5687	143	0.1	2.7	0.1	0.1	0.1	29.6	96	4	0.1	2.3
	11	8.3	2	41	5809	122	0.1	1.6	0.1	0.1	0.1	30.1	96.3	3.3	0.3	3.3
	14	8.35	2	13	5840	115	0.1	2.6	0.1	0.1	0.1	30.1	96.8	3.1	0.1	2.3
	6B	8.4	2	7	5663	99	0.1	3.7	0.1	0.1	0.1	29.1	97.2	2.8	0.1	1.7
Bottom 10 cm	1	8.32	2	34	5587	179	0.1	2	0.1	0.1	0.1	29.4	94.7	5	0.3	2.9
	2	8.27	2	22	5543	223	0.1	1.7	0.1	0.2	0.1	29.5	93.6	6.2	0.2	2.5
	3	8.4	2	23	5420	172	0.2	2.5	0.1	0.6	0.2	28.5	94.8	5	0.2	3.7
	4	8.32	2	28	5749	140	0.1	2.1	0.1	0.1	0.1	29.9	95.9	3.9	0.2	3.3
	5	8.48	2	26	5651	136	0.1	3.2	0.1	0.1	0.1	29.4	96	3.8	0.2	2.4
	6	8.52	2	21	5579	175	0.1	2.1	0.1	0.1	0.1	29.3	94.9	4.9	0.2	3.9
	7	8.34	2	40	5602	133	0.1	2.2	0.1	0.1	0.1	29.1	95.9	3.8	0.4	4.2
	8	8.48	2	34	5621	165	0.1	2.7	0.1	0.1	0.1	29.5	95.1	4.6	0.3	3.2
	9	8.31	2	22	5594	105	0.1	3	0.1	0.2	0.1	28.8	96.8	3	0.2	2.5
	10	8.52	2	24	5696	196	0.1	2.6	0.1	0.1	0.1	30.1	94.5	5.4	0.2	3
	11	8.41	2	48	5806	141	0.1	1.7	0.1	0.2	0.1	30.2	95.8	3.8	0.4	3.1
	14	8.39	2	20	5536	156	0.1	2.3	0.1	0.1	0.1	28.9	95.4	4.4	0.2	4.3
	6B	8.48	2	32	5596	165	0.1	2.3	0.1	0.1	0.1	29.4	95.1	4.6	0.3	3.8

Appendix D - Water quality data, Spring controlled flood event. nm – Not measured; n.a. – Not applicable.

Location	Event Time (min)	Sample ID	PO_4-P µg/L	NH_3-N µg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP µg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
Inlet	2	1	3.8	49.1	2.34	1.28	nm	nm	1.98	0.07	3.59	135.7	n.a.
Inlet	32	2	9.5	38.3	2.14	1.43	nm	nm	2.03	0.09	3.49	113.3	n.a.
Inlet	62	3	11.4	39.1	2.12	1.76	nm	nm	2.13	0.07	3.34	86.7	n.a.
Inlet	92	4	10.3	48.9	2.36	1.67	nm	nm	2.05	0.07	2.65	65.3	n.a.
Inlet	122	5	15.8	37.4	1.69	1.54	nm	nm	2.07	0.06	3.31	54.0	3.2
Inlet	152	6	19.5	42.9	2.33	1.49	nm	nm	2.03	0.07	3.20	51.3	7.1
Inlet	182	7	11.1	48.0	2.42	1.40	nm	nm	2.01	0.07	3.40	60.8	5.9
Inlet	212	8	22.0	31.5	2.41	1.75	nm	nm	2.05	0.07	3.43	69.3	5.9
Inlet	242	9	11.3	46.9	2.44	1.68	nm	nm	2.06	0.05	3.32	58.8	5.5
Inlet	272	10	14.7	31.8	1.83	1.55	nm	nm	2.04	0.06	3.05	51.7	5.9
Inlet	302	11	24.6	34.7	2.39	2.11	nm	nm	2.03	0.08	1.88	73.8	11.1
Inlet	332	12	7.1	32.2	1.72	1.28	nm	nm	2.01	0.04	2.91	39.0	19.0
Inlet	362	13	13.0	34.5	1.87	1.34	nm	nm	2.02	0.06	2.95	41.2	n.a.
Inlet	392	14	15.1	23.5	2.22	1.24	nm	nm	2.01	0.06	2.77	36.2	n.a.

			PO_4-P	NH_3-N	NO_3/NO_2-N	DOC	TOC	TDP	TDN	TP	TN	TSS	Br
Location	**Event Time (min)**	**Sample ID**	**μg/L**	**μg/L**	**mg/L**	**mg/L**	**mg/L**	**μg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**
Outlet	8	1	9.2	16.5	0.19	9.36	nm	nm	0.40	0.08	0.72	52.5	0.5
Outlet	38	2	11.5	22.5	1.75	2.79	nm	nm	1.66	0.09	2.41	67.2	0.6
Outlet	68	3	16.2	14.4	2.29	1.79	nm	nm	1.92	0.04	2.64	33.0	0.5
Outlet	98	4	18.5	29.6	2.34	1.76	nm	nm	1.98	0.06	3.11	30.9	0.5
Outlet	128	5	12.6	23.1	2.34	1.64	nm	nm	2.01	0.06	3.13	22.0	0.6
Outlet	188	6	14.9	18.2	2.29	1.73	nm	nm	1.97	0.05	2.93	14.8	5.0
Outlet	248	7	8.6	17.4	2.32	1.39	nm	nm	1.98	0.05	2.81	17.6	6.6
Outlet	308	8	17.2	20.7	2.20	1.54	nm	nm	1.97	0.06	2.91	21.4	7.0
Outlet	368	9	10.6	10.9	2.30	1.79	nm	nm	1.98	0.04	2.70	14.6	2.9
Outlet	428	10	13.8	14.5	2.25	2.52	nm	nm	1.89	0.04	2.04	8.4	1.5
Outlet	488	11	13.4	11.3	2.25	1.77	nm	nm	1.83	0.03	2.57	6.2	1.4
Outlet	500	12	7.1	11.2	2.33	1.65	nm	nm	1.82	0.04	2.38	9.3	1.5
1	1		13.9	35.3	1.88	1.95	nm	nm	1.65	0.07	2.70	61.0	n.a.
2	1		18.7	35.9	1.95	1.98	nm	nm	1.63	0.08	2.90	60.4	n.a.
3	1		14.4	32.5	1.96	1.83	nm	nm	1.56	0.07	2.68	67.2	n.a.

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
4	1		12.5	41.6	2.30	2.40	nm	nm	1.77	0.07	2.45	32.0	n.a.
5	1		13.6	35.6	2.43	2.08	nm	nm	1.91	0.07	1.84	45.8	n.a.
6	1		13.8	19.2	1.70	4.60	nm	nm	1.32	0.08	2.16	30.2	n.a.
8	1		12.5	24.0	2.09	1.87	nm	nm	1.81	0.07	1.24	33.0	n.a.
9	1		11.3	16.5	2.13	1.75	nm	nm	1.64	0.06	2.49	30.8	n.a.
10	1		10.7	11.6	1.17	2.69	nm	nm	1.63	0.06	2.45	28.7	n.a.
11	1		12.7	42.7	2.27	1.59	nm	nm	1.91	0.06	2.74	60.6	n.a.
12	1		9.6	12.8	2.21	2.04	nm	nm	1.80	0.07	2.56	26.4	n.a.
1	2		16.1	26.5	2.14	1.30	nm	nm	1.67	0.07	2.66	40.0	4.8
2	2		14.1	29.8	2.42	1.10	nm	nm	1.18	0.07	2.01	48.4	6.7
3	2		16.9	26.9	2.34	1.58	nm	nm	1.89	0.06	2.46	27.0	4.8
4	2		11.2	29.9	2.33	1.70	nm	nm	1.81	0.06	2.18	16.3	1.7
5	2		14.2	25.6	2.23	1.39	nm	nm	1.82	0.06	2.39	27.3	5.7
6	2		11.0	26.0	1.95	2.96	nm	nm	1.64	0.07	2.58	14.4	0.5
8	2		18.4	25.2	2.52	1.70	nm	nm	1.91	0.06	2.42	18.8	2.7

Location	Event Time (min)	Sample ID	PO_4-P µg/L	NH_3-N µg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP µg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
9	2		7.3	18.7	2.44	1.27	nm	nm	1.49	0.05	2.15	23.0	4.8
10	2		13.3	12.9	1.75	2.06	nm	nm	1.36	0.06	2.23	11.8	0.6
11	2		13.3	22.1	1.98	1.34	nm	nm	1.87	0.07	1.41	55.0	5.6
12	2		12.0	14.8	2.03	1.74	nm	nm	1.83	0.06	2.43	18.7	3.1
1	3		9.2	22.0	2.05	1.32	nm	nm	1.90	0.05	1.68	45.8	5.6
2	3		5.6	23.3	1.43	1.36	nm	nm	1.88	0.04	2.10	81.0	6.6
3	3		12.8	24.3	2.13	1.34	nm	nm	1.77	0.06	1.57	32.8	7.5
4	3		14.4	20.0	1.57	1.60	nm	nm	1.87	0.06	1.72	13.1	6.9
5	3		19.4	19.3	2.50	1.21	nm	nm	1.77	0.07	2.06	40.6	6.6
6	3		17.5	15.0	2.21	2.73	nm	nm	1.66	0.06	1.82	12.1	5.5
8	3		11.7	30.8	2.27	1.82	nm	nm	1.89	0.07	2.20	22.2	7.0
9	3		10.9	18.8	1.98	1.56	nm	nm	1.83	0.06	1.88	23.0	7.1
10	3		15.3	12.0	2.12	2.91	nm	nm	1.74	0.03	1.20	10.6	5.9
11	3		6.1	22.9	2.42	1.25	nm	nm	1.67	0.05	1.55	38.7	7.2
12	3		9.7	13.9	2.27	1.30	nm	nm	1.68	0.06	1.53	18.2	5.9

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
1	4		8.1	23.4	2.19	1.34	nm	nm	1.93	0.06	2.20	24.6	0.8
2	4		15.7	24.0	2.00	1.22	nm	nm	1.80	0.06	2.20	38.6	0.6
3	4		14.6	22.3	2.42	1.22	nm	nm	1.75	0.05	1.91	16.1	1.3
4	4		11.2	21.7	2.50	1.29	nm	nm	1.77	0.04	1.83	14.0	4.3
5	4		17.4	21.4	2.40	1.02	nm	nm	1.44	0.06	1.95	59.0	0.9
6	4		12.0	24.2	2.31	1.90	nm	nm	1.64	0.05	1.69	15.7	6.8
8	4		6.6	23.5	2.21	1.76	nm	nm	1.82	0.04	2.20	15.4	2.5
9	4		13.9	15.7	2.11	1.62	nm	nm	1.82	0.05	1.59	14.9	1.2
10	4		7.9	10.7	2.31	1.80	nm	nm	1.75	0.04	1.96	7.4	5.0
11	4		10.0	26.6	2.38	1.07	nm	nm	1.59	0.06	1.06	38.3	0.0
12	4		6.8	13.9	1.81	1.34	nm	nm	1.90	0.05	1.72	11.9	2.2
1		syn	10.8	29.7	2.41	1.44	nm	nm	1.49	0.08	1.80	nm	6.3
2		syn	15.8	29.8	1.85	1.49	nm	nm	1.55	0.07	1.88	nm	5.8
3		syn	15.7	26.3	1.60	1.74	nm	nm	1.38	0.07	2.18	nm	6.4
4		syn	13.5	28.8	2.25	1.34	nm	nm	1.95	0.07	1.64	nm	6.3

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
5		syn	12.2	23.4	1.82	1.51	nm	nm	1.86	0.06	1.63	nm	6.5
6		syn	12.5	23.3	2.02	1.29	nm	nm	1.96	0.07	2.33	nm	5.1
7		syn	15.6	22.3	1.91	1.93	nm	nm	1.77	0.05	1.78	nm	5.0
8		syn	11.1	35.3	2.29	1.48	nm	nm	1.93	0.05	1.23	nm	6.5
9		syn	10.6	19.9	2.30	1.46	nm	nm	1.94	0.06	2.38	nm	6.4
10		syn	9.6	19.0	2.04	1.85	nm	nm	1.74	0.05	2.31	nm	3.5
11		syn	13.7	28.5	1.84	1.41	nm	nm	1.90	0.05	2.64	nm	5.2
12		syn	23.3	11.8	2.14	2.35	nm	nm	1.76	0.05	2.39	nm	6.2
13		syn	7.7	16.4	2.08	1.99	nm	nm	1.72	0.04	1.93	nm	3.6
14		syn	8.0	12.9	1.48	2.55	nm	nm	1.64	0.05	2.51	nm	2.6
15		syn	25.5	18.4	1.35	5.81	nm	nm	1.34	0.08	2.24	nm	1.0
16		syn	17.0	18.4	0.33	13.69	nm	nm	0.83	0.20	2.23	nm	0.7
17		syn	15.0	18.5	1.33	1.90	nm	nm	1.81	0.06	2.54	nm	4.7
18		syn	15.2	17.4	2.03	3.28	nm	nm	1.39	0.06	1.59	nm	2.2
19		syn	18.2	29.4	2.26	1.71	nm	nm	1.61	0.07	2.48	nm	5.4

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
20		syn	14.6	26.6	2.28	1.58	nm	nm	1.49	0.06	2.51	nm	5.4
21		syn	13.3	22.5	2.11	1.86	nm	nm	1.89	0.05	2.22	nm	3.3
22		syn	8.8	19.7	1.91	1.61	nm	nm	1.90	0.05	2.54	nm	6.3
23		syn	15.4	22.9	1.45	1.90	nm	nm	0.70	0.10	2.55	nm	1.2
24		syn	15.3	22.2	1.62		nm	nm		0.06	1.99	nm	4.8
25		syn	32.7	20.6	1.86		nm	nm		0.06	1.68	nm	3.2
Outlet		FF1	14.6	30.2	0.22	8.69	nm	nm	0.52	0.07	0.43	nm	nm
Outlet		FF2	11.9	15.5	0.38	8.34	nm	nm	0.59	0.08	0.44	nm	nm
Outlet		FF3	12.5	13.8	0.83	6.10	nm	nm	0.74	0.09	0.97	nm	nm
Outlet		FF4	21.7	14.5	1.15	5.59	nm	nm	1.02	0.09	1.34	nm	nm
Outlet		FF5	9.5	14.2	1.52	4.36	nm	nm	1.32	0.09	1.56	nm	nm
Outlet		FF6	16.7	30.3	1.77	3.52	nm	nm	1.43	0.09	1.73	nm	nm
stream		SS	12.3	61.6	2.38	1.72	nm	nm	1.86	0.05	2.30	nm	nm
stream		BG	15.9	47.0	2.52	1.60	nm	nm	1.91	0.02	3.01	12.5	
Pool 1		P1	15.1	33.6	2.19	1.48	nm	nm	1.71	0.03	2.28	4.0	1.0

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
Pool 2		P2	5.8	23.8	2.02	1.75	nm	nm	1.60	0.03	2.19	6.8	2.3
Lo Marsh		LM	4.8	17.2	1.09	1.89	nm	nm	0.83	0.02	0.93	2.1	1.2
Stream		SW1	8.8	0.0	2.11	nm	nm	nm	nm	0.03	1.76	nm	nm
Stream		SW2	7.2	0.0	2.41	nm	nm	nm	nm	0.04	2.49	nm	nm
Stream		SW3	12.4	0.0	2.27	nm	nm	nm	nm	0.03	2.27	nm	nm
Stream		SW4	12.2	0.0	2.38	nm	nm	nm	nm	0.03	2.47	nm	nm
Stream		SW5	6.5	0.0	2.14	nm	nm	nm	nm	0.03	2.74	nm	nm
Stream		SW6	5.6	0.0	2.24	nm	nm	nm	nm	0.03	1.91	nm	nm
Stream		SW7	4.0	0.0	2.22	nm	nm	nm	nm	0.03	2.43	nm	nm

Appendix E - Water quality data, Fall controlled flood event. nm – Not measured; n.a. – Not applicable; bd – Below detection.

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
Inlet	10	2	11.0	52.4	2.97	2.50	5.30	8.45	3.20	0.22	3.96	10.2	n.a.
Inlet	30	3	0.8	19.2	2.81	1.50	2.50	bd	3.08	0.07	3.33	87.7	n.a.
Inlet	50	4	2.7	31.6	2.90	1.60	2.30	27.65	3.11	0.05	3.29	63.0	n.a.
Inlet	70	5	7.1	27.4	2.83	1.50	1.60	bd	3.11	0.05	3.21	49.5	n.a.
Inlet	90	6	6.5	32.0	2.82	1.30	2.00	bd	3.10	0.04	3.14	40.3	n.a.
Inlet	110	7	6.8	19.7	2.79	1.20	2.50	28.55	3.01	0.06	3.24	102.0	n.a.
Inlet	130	8	3.2	14.0	2.76	1.90	3.80	bd	3.07	0.08	3.41	147.7	n.a.
Inlet	150	9	5.6	33.3	2.80	1.70	1.60	bd	3.16	0.06	3.41	73.3	n.a.
Inlet	170	10	5.9	22.3	2.95	1.60	1.60	10.25	3.08	0.05	3.35	53.0	n.a.
Inlet	186	11	5.2	20.8	2.73	2.10	3.20	20.75	2.96	0.05	3.11	83.9	n.a.
Inlet	225	12	3.8	18.6	2.81	1.10	1.30	7.40	2.99	0.05	3.14	48.3	n.a.
Inlet	255	13	4.6	9.4	2.79	1.20	1.90	25.70	2.90	0.05	3.05	46.6	n.a.
Inlet	285	14	3.1	21.1	2.76	1.10	1.30	bd	3.07	0.05	2.99	41.1	n.a.
Inlet	315	15	5.2	31.3	2.86	1.20	1.40	bd	2.99	0.04	3.14	46.0	1.4

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
Inlet	345	16	6.6	27.6	2.93	1.40	1.50	bd	3.02	0.04	3.15	34.2	n.a.
Inlet	375	17	2.6	13.5	2.93	1.80	1.70	6.20	3.04	0.03	3.05	28.0	n.a.
Inlet	405	18	4.1	30.1	2.75	1.40	2.10	bd	3.01	0.04	3.09	20.6	10.0
Inlet	435	19	3.9	33.1	2.73	1.40	1.40	bd	3.04	0.09	3.11	37.0	8.5
Inlet	465	20	3.9	23.9	2.79	1.50	2.20	bd	3.01	0.04	3.09	22.7	5.1
Outlet	38	0.5	8.7	28.7	1.69	8.20	11.20	14.45	2.06	0.16	2.73	213.6	n.a.
Outlet	39	1	4.4	10.0	2.16	2.60	9.80	bd	2.72	0.22	3.05	120.7	n.a.
Outlet	49	2	4.4	14.3	2.08	5.20	7.60	bd	2.33	0.14	2.93	174.0	n.a.
Outlet	59	3	3.9	10.5	2.51	4.00	4.20	bd	2.59	0.09	2.96	38.5	n.a.
Outlet	69	4	7.6	15.5	2.54	2.80	2.70	bd	2.84	0.07	2.97	27.5	n.a.
Outlet	79	5	5.9	9.9	2.69	2.50	2.50	bd	2.89	0.14	3.00	41.3	n.a.
Outlet	89	6	3.2	21.4	2.71	2.30	2.90	bd	3.01	0.06	3.12	21.0	n.a.
Outlet	99	7	7.5	28.0	2.70	1.90	1.90	23.45	2.95	0.05	3.09	16.0	n.a.
Outlet	109	8	3.7	15.4	2.70	1.80	1.90	bd	2.90	0.09	3.00	13.0	n.a.
Outlet	119	9	2.7	20.1	2.73	1.80	1.90	bd	3.08	0.04	3.08	9.0	n.a.

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
Outlet	129	10	8.0	18.4	2.76	1.60	2.20	bd	2.98	0.05	3.12	9.8	n.a.
Outlet	139	11	5.0	15.9	2.89	1.70	1.60	bd	3.02	0.04	3.20	8.6	n.a.
Outlet	149	12	6.4	9.5	2.70	1.70	2.30	bd	2.90	0.03	3.03	15.5	n.a.
Outlet	159	13	3.0	10.6	2.79	1.50	2.30	20.45	2.89	0.05	3.12	23.8	n.a.
Outlet	179	14	5.2	16.9	2.86	1.70	2.40	bd	2.99	0.03	3.12	24.3	n.a.
Outlet	209	15	4.6	15.1	2.83	1.60	1.60	bd	3.10	0.04	3.15	17.3	n.a.
Outlet	239	16	3.7	14.4	2.80	1.50	1.60	bd	3.14	0.04	3.20	11.8	n.a.
Outlet	259	17	2.3	14.4	2.70	2.00	2.90	bd	2.92	0.04	3.24	8.0	n.a.
Outlet	289	18	4.6	11.0	2.70	1.40	1.50	bd	2.93	0.05	2.97	7.6	n.a.
Outlet	319	19	4.1	9.4	2.82	1.50	1.50	bd	2.92	0.03	3.00	7.0	n.a.
Outlet	349	20	3.9	10.5	2.78	1.60	1.60	bd	2.96	0.03	2.99	5.5	n.a.
Outlet	379	21	3.8	9.2	2.79	1.60	1.60	bd	2.98	0.03	3.06	6.3	1.7
Outlet	409	22	2.9	10.8	2.68	1.60	2.50	bd	3.17	0.02	3.06	6.9	8.2
Outlet	439	23	4.6	10.1	2.72	1.50	1.50	bd	2.93	0.02	3.23	5.9	8.1
Outlet	469	24	4.0	14.6	2.73	1.40	1.50	23.45	3.16	0.02	3.02	6.4	8.8

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
1	1		7.0	35.0	2.91	nm	nm	6.20	3.07	0.04	3.11	27.4	n.a.
2	1		6.6	25.0	2.91	nm	nm	bd	2.24	0.07	3.23	13.3	n.a.
3	1		4.9	27.9	2.91	nm	nm	bd	2.71	0.05	2.24	64.6	n.a.
4	1		11.2	24.5	2.88	nm	nm	55.25	3.10	0.09	3.17	17.0	n.a.
5	1		5.3	18.9	2.85	nm	nm	bd	2.75	0.03	3.08	11.5	n.a.
6	1		6.3	15.6	2.63	nm	nm	19.70	1.81	0.20	3.51	65.3	n.a.
8	1		6.7	23.4	1.90	nm	nm	21.50	2.89	0.08	2.96	20.0	n.a.
9	1		3.2	28.1	2.82	nm	nm	bd	1.55	0.03	3.02	11.6	n.a.
10	1		14.9	17.7	2.83	nm	nm	66.50	3.17	0.12	3.32	20.2	n.a.
11	1		4.7	23.6	2.80	nm	nm	bd	2.96	0.03	2.91	25.2	n.a.
12	1		5.7	23.2	1.89	nm	nm	bd	2.56	0.05	3.00	16.2	n.a.
13	1		3.8	27.6	2.78	nm	nm	38.45	2.78	0.03	3.08	9.1	n.a.
14	1		4.6	29.2	2.81	nm	nm	bd	1.43	0.04	3.05	9.9	n.a.
1	2		7.3	15.4	2.70	2.00	1.90	15.65	2.84	0.04	3.11	16.0	7.6
2	2		4.5	31.8	2.78	1.70	1.80	13.40	0.95	0.03	3.35	22.0	8.2

Location	Event Time (min)	Sample ID	PO_4-P µg/L	NH_3-N µg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP µg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
3	2		2.3	17.0	2.50	1.70	2.40	30.35	2.87	0.04	2.16	26.6	2.7
4	2		6.6	14.7	1.20	2.60	2.20	38.45	2.81	0.08	2.45	9.8	0.5
5	2		3.7	20.2	2.86	1.80	1.70	bd	2.68	0.02	3.08	10.3	6.8
6	2		3.4	13.2	2.60	3.60	6.20	10.55	2.86	0.27	3.68	356.7	0.0
8	2		4.9	15.0	2.66	2.20	2.10	24.35	2.80	0.04	3.06	6.0	0.0
9	2		4.1	27.5	1.81	1.60	2.10	bd	2.89	bd	3.24	16.9	1.6
10	2		5.7	20.4	2.77	1.80	2.50	27.50	2.93	0.05	3.11	14.1	0.0
11	2		7.2	36.0	2.92	1.20	1.50	bd	0.99	0.04	3.29	21.9	12.1
12	2		2.5	21.0	2.60	nm	nm	bd	2.89	bd	2.33	19.4	8.1
13	2		2.4	31.2	2.79	nm	nm	bd	1.03	0.03	3.20	74.6	8.1
14	2		3.2	24.7	2.85	1.50	2.20	bd	2.92	0.03	3.11	7.6	6.9
1	3		6.9	39.5	2.85	2.00	2.00	6.35	0.98	0.03	2.81	20.6	9.2
2	3		3.7	29.0	2.83	1.50	1.80	bd	1.07	0.03	3.18	12.6	9.6
3	3		4.6	27.9	2.78	1.70	2.20	bd	0.91	0.05	3.03	10.6	9.0
4	3		9.6	17.8	2.82	1.30	2.90	33.20	2.89	0.07	3.12	9.9	0.8

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
5	3		3.6	31.8	2.83	1.30	1.70	bd	3.13	0.03	3.09	9.2	8.1
6	3		5.1	13.3	2.79	1.90	4.00	bd	2.77	0.17	2.22	134.5	4.0
8	3		3.8	14.3	2.63	2.00	2.70	14.75	3.02	0.04	3.08	4.2	1.5
9	3		3.0	30.4	2.84	1.90	1.90	bd	2.95	0.02	2.76	9.2	9.0
10	3		8.1	13.2	2.62	2.70	2.10	11.45	2.15	0.05	3.03	20.1	2.5
11	3		6.9	37.5	3.01	2.20	1.60	8.00	3.05	0.04	3.38	16.4	12.6
12	3		4.9	37.3	2.86	nm	nm	28.70	2.92	0.03	2.91	31.8	9.6
13	3		4.4	33.2	2.85	nm	nm	27.20	2.89	0.03	3.20	14.9	8.7
14	3		3.8	31.7	2.96	1.40	1.80	5.90	2.92	0.04	3.12	7.9	8.7
1	4		4.5	39.4	2.85	1.20	1.50	34.10	2.90	0.04	3.08	12.3	9.5
2	4		5.8	35.4	2.87	1.60	1.40	bd	2.95	0.02	2.90	11.4	9.5
3	4		6.2	35.8	2.79	1.70	1.80	40.40	2.78	0.03	3.09	10.5	9.5
4	4		8.8	20.4	2.77	2.40	2.10	33.35	2.98	0.04	3.09	4.7	4.1
5	4		4.4	32.9	2.83	1.80	1.50	11.60	2.90	0.03	2.43	10.0	8.4
6	4		13.6	36.7	2.67	3.80	8.60	28.40	2.69	0.14	2.51	183.0	4.5

Location	Event Time (min)	Sample ID	PO_4-P μg/L	NH_3-N μg/L	NO_3/NO_2-N mg/L	DOC mg/L	TOC mg/L	TDP μg/L	TDN mg/L	TP mg/L	TN mg/L	TSS mg/L	Br mg/L
8	4		11.5	24.7	2.77	3.50	2.10	19.25	2.84	0.06	3.09	4.0	5.2
9	4		3.2	27.9	2.79	1.30	2.00	bd	2.87	0.03	3.24	17.2	7.9
10	4		6.6	29.6	2.72	2.50	2.20	20.90	2.77	0.05	3.12	14.5	5.1
11	4		4.9	39.8	2.94	1.20	1.80	bd	2.98	0.03	3.17	15.9	11.5
12	4		3.9	40.1	2.78	nm	nm	bd	2.90	0.01	3.12	11.0	4.6
13	4		4.0	33.7	2.79	nm	nm	bd	2.81	bd	3.08	40.5	9.5
14	4		5.6	31.9	2.98	1.90	1.70	8.60	2.90	0.03	3.03	6.3	9.1

Appendix F - Nutrient data summary table of event inlet and outlet average constituent concentrations; range of concentrations in parentheses. nm – Not measured.

		PO_4 µg/L	NH_3 µg/L	NO_3 mg/L	DOC mg/L	TOC mg/L	TDP µg/L	TDN mg/L	TP µg/L	TN mg/L	TSS mg/L
Fall	Inlet	4.9 (1-11)	25.3 (9-52)	2.8 (2.7-3.0)	1.5 (1.1-2.5)	2.2 (1.3-5.3)	16.9 (6-29)	3.0 (2.9-3.2)	60.8 (33-217)	3.2 (3.0-4.0)	54.5 (10-147)
Fall	Outlet	4.7 (2-9)	14.6 (9-28)	2.6 (1.7-2.9)	2.3 (1.4-8.2)	3.0 (1.5-11.2)	20.4 (14-23)	2.9 (2.1-3.2)	62.6 (22-223)	3.1 (2.7-3.2)	33.6 (6-214)
Spring	Inlet	13.5 (4-25)	38.5 (24-49)	2.2 (1.7-2.4)	1.5 (1.2-2.1)	nm	nm	2.0 (2.0-2.1)	65.2 (36-93)	3.1 (1.9-3.6)	67.0 (36-135)
Spring	Outlet	12.8 (7-19)	17.5 (11-30)	2.1 (0.2-2.3)	2.5 (1.4-9.4)	nm	nm	1.8 (0.4-2.0)	54.5 (34-90)	2.5 (0.7-3.1)	24.8 (6-67)

Appendix G - Timeline of field activities during spring controlled flood event.

Field Log - Fall AOE	**11/12/08**

Time	**Activity**
8:00	Background creek samples collected
9:20	Inflow pump on, Nutrient injection pump on
9:22	Inflow occurring
	Inlet ISCO program - Sample every 5 min, composites of 20 min
	Outlet ISCO program - Sample every 2 min, composites of 10 min
9:53	Outfall at outlet
11:15	Increased pump rate
12:16	Stage steady-state declared
	ISCO program change
	Inlet ISCO program - Sample every 5 min, composites of 30 min
	Outlet ISCO program - Sample every 5 min, composites of 30 min
12:36	Dye slug added
13:00	Dye reaches outlet
13:30	Snapshot-grab sample 1
14:54	KBr Tracer pump on
	Transect A - Sample every 3 min
	Transect B - Sample every 5 min, 5 min delayed start
	Transect C - Sample every 5 min, 10 min delayed start
15:24	Transect A - Sample every 15 min
15:27	Snapshot-grab sample 2
15:54	Transect B - Sample every 15 min
16:00	ADV Velocity measurements begin (lasting until 17:00)
16:10	Snapshot-grab sample 3
16:24	Transect C - Sample every 15 min
16:55	Snapshot-grab sample 4
17:01	Inflow pump off

Appendix H - Timeline of field activities during spring controlled flood event.

Field Log - Spring	**5/19/09**

Time	**Activity**
8:00	Data sonde and temperature probe deployment
9:33	Background creek samples collected
10:25	Inflow pump on, Inflow into wetland, Nutrient injection pump on
	Inlet ISCO not functioning, Inlet grab samples collected every 30 min
	Outlet ISCO not sampling correctly, Outlet grab samples collected every30 min
10:33	Diffuser adjustment
10:48	Outfall at outlet, first flush samples collected
12:00	Snapshot-grab sample 1
12:30	Stage steady-state declared
12:32	KBr Tracer injection pump on
	Transect A - Sample every 3 min
	Transect B - Sample every 5 min
	Transect C - Sample every 5 min, 15 min delayed start
12:48	Outlet grab samples collected every hour
13:12	Transect A - Sample every 10 min
1:53	Transect B - Sample every 15 min
14:13	Transect C - Sample every 20 min
13:25	Snapshot-grab sample 2
13:50	Intake fluctuation due to whirlpools, Decreased pump rate
14:15	Synoptic sample (Lasting 15 min)
15:06	Major pump fluctuation, Intake hose adjustment
15:35	Snapshot-grab sample 3
16:06	KBr Tracer injection pump off
16:55	Inflow pump off

Appendix I - Photographs of soil core data from cores collected in the adjacent hillslope (left) and wetland center (right). Photo taken by Andrea Ludwig.

Appendix J - Photographic documentary prepared for the Opequon Targeted Watershed Grant Final Report in June 2010.

Hedgebrook Farm Constructed Floodplain Wetland
Photographic Documentary

Winchester, VA
Implemented May 2007

Stormwater best management practice for treatment of overbank creek flows and demonstration site for experimental flooding events for nutrient removal studies. Photo taken by Andrea Ludwig.

May 2007: Site before construction. Opequon Creek running in wooded corridor on right. Wetland grasses in center. Fescue pastureland. Photo taken by Andrea Ludwig.

May 2007: Completed excavation of approximately 650 yd^3 of material. Removal of fescue. Footprint is approximately 2040 m^2. Photo taken by Andrea Ludwig.

July 2007: Installation of outlet structure, 1-ft H-flume. Inlet and outlet channels cut. Annual rye being watered as cover crop until wetland planting in November 2007. Photo taken by Andrea Ludwig.

July 2007: Inlet channel being excavated. Opequon creek in foreground. Photo taken by Andrea Ludwig.

January 2008: Nested piezometers installed throughout wetland to monitor water table and hydraulic gradients. Five nests of three piezometers monitor water levels in different soil layers. Photo taken by Andrea Ludwig.

Spring 2008: Surveying of piezometer locations. Forebay in foreground. Outlet automated ISCO sampler in background. Inlet and outlet instrumented with ISCOs. Each piezometer instrumented with continuous water level loggers. Photo taken by Andrea Ludwig.

November 2008: Fall controlled flooding event for data collection. Nutrient amendments, tracer, and dye injected to measure residence time and nutrient removal capacity. Looking upstream from outlet flume. Three sampling transects are suspended over the wetland. Photo taken by Andrea Ludwig.

December 2008: Dormant vegetation in the winter. Looking upstream from Exit pool.
Photo taken by Andrea Ludwig.

May 2009: Spring season controlled flooding event. Looking from the inlet channel in the right foreground. ISCO sampler in the center. Suspended sampling transects. Photo taken by Andrea Ludwig.

May 2009: Inlet channel during controlled flooding event. Six-inch trash pump pumps creek water through wetland and small injection pump adds tracer and low-level nutrient amendments for experiment. Photo taken by Andrea Ludwig.

June 2009: Second year vegetation grow-out. Photo taken by Andrea Ludwig.

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